

LONG-TERM IMPACTS OF FUEL TREATMENTS ON TREE GROWTH AND
ABOVEGROUND BIOMASS ACCUMULATION IN PONDEROSA PINE FORESTS OF
THE NORTHERN ROCKY MOUNTAINS.

By

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Long-term impacts of fuel treatments on tree growth and aboveground biomass accumulation in ponderosa pine forests of the Northern Rocky Mountains.

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In western North America, many low-elevation, dry forest types historically experienced frequent, low-severity fires. However, European settlement and fire suppression policies have contributed to over a century of fire exclusion, substantially altering forest structure and composition. There is considerable interest in restoring fire resilient characteristics to these forests through fuel reduction treatments. One limitation of current research on the impacts of fuel treatments is treatment longevity, as few studies have been able to quantify long-term responses to commonly applied treatments. This research evaluated tree growth and aboveground biomass responses 23 years after treatment in two silvicultural installations with different underburning prescriptions. Thinning and shelterwood treatments were implemented in 1991 in the Lick Creek drainage of southwestern Montana. Aside from a no-cut control, three post-harvest burning prescriptions were applied in each installation: a no burn, a spring/wet burn, and a fall/dry burn. In 2015 stand density was lower in all treated stands relative to the control, and peak growth of volume accumulation had passed. Stand-level basal area increment was the same across treatments, while tree-level basal area increment was greater in the fuel treatments. In the thinning live tree biomass recovered to pre-harvest levels by 2005 in all three fuel treatments, but was still less than the control. Forest floor biomass was lower in the two burned treatments relative to the two unburned treatments. In the shelterwood, tree biomass had recovered to pre-harvest levels in all fuel treatments by 2015, and was lower in the two burned treatments relative to the two unburned treatments. Forest floor biomass also tended to be lower in the burned treatments. This research suggests that tree biomass in fuel treatments can recover to pre-harvest levels within as little as 10 years while still maintaining reduced stand densities that advance several restoration objectives. Additionally, burning treatments maintain reduced forest floor biomass, even 23 years after treatment, indicating a persistent legacy of burning on this component. However, high regeneration densities indicate that a treatment regime strategies that includes understory treatments are required across treatments to maintain structures conducive to low-severity fire.

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SECTION 1:

Aboveground biomass responses to fuel treatments in fire-frequent forests of the American West:
a review

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Chair of the Supervisory Committee:
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Abstract

In western North America, many low-elevation, dry forest types historically experienced frequent, low-severity fires. However, European settlement and fire suppression policies have contributed to over a century of fire exclusion, substantially altering forest structure and composition. Specifically, many forests have experienced considerable increases in forest biomass relative to pre-settlement conditions. As a result, carbon storage in these forests has increased, which has partially offset rises in anthropogenic carbon emissions over the past century. However, it is becoming increasingly difficult to exclude fire from the landscape, as changes in climate are resulting in longer and drier summers, and an excess of forest fuels are present across much of the landscape. High-severity fires are increasing in frequency, magnitude, and duration, resulting in substantial carbon offset reversals. Fuel treatments aim to restore low-severity fire conditions by strategically reducing available forest fuels. However, it is unclear as

to how different fuel treatments impact aboveground biomass, especially in the long term. This review compiled existing research on the effects of fuel treatments on total aboveground biomass, as well as both live and dead tree biomass, vegetative biomass, and forest floor biomass. In the short-term total aboveground and live tree biomass unanimously decreased after fuel reduction treatments, while responses of vegetative biomass were highly dependent on species and treatment type. Forest floor biomass decreased in treatments that included broadcast burning, and were variable in thin-only treatments. From compilation of existing literature, two knowledge-gaps were identified. The first gap was geographic location, as no studies have examined biomass responses to fuel treatments in the northern Rocky Mountains. The second was longevity, as few studies evaluated biomass responses more than 3 years after treatment. From this review, we concluded that more research is needed on longer term responses across a range of fire-frequent forest types in the western United States.

1.1 Introduction

In the western United States, wildfires are increasing in magnitude, intensity, and severity. In 2015, wildfires burned more than 10 million acres in the U.S., primarily in Alaska and the western states (National Interagency Fire Center 2016a). Since the National Interagency Fire Center began collecting data in 1960, only six other years have surpassed 8 million acres burned, all occurring since 2004, and only three others have exceeded 9 million acres. The escalation of large fires over an alarmingly short period of time is raising concerns for current and future conditions of western forests. Human intervention is considered a significant contribution to this augmentation of wildfires, both directly and indirectly. European settlement drastically altered disturbance regimes through land use changes and fire suppression policies, while increases in human populations living and recreating in the wildland urban interface

(WUI) has substantially increased ignition probability (Pyne 1982). Anthropogenic contributions to surges in global atmospheric carbon and resulting climate change are predicted to extend wildfire seasons, increase fire severity and intensify occurrence of extreme fire weather (Westerling et al. 2006, Parry et al. 2007, Walsh 2008, Jolly et al. 2015). With approximately 323 million people living in the United States (United States Census Bureau 2016), wildfires will never be allowed to return to pre-settlement circumstances (United States Census Bureau 2016). As a result, increased land management activities will be required to simulate effects of disturbance, especially where wildfires cannot burn without impairing human safety.

1.2 Historical fire regimes of the West

Low-elevation, dry, ponderosa pine and mixed-conifer forests are among the ecosystems in western North America currently experiencing increases in wildfire severity and extent. Historically, the disturbance regime of these forests entailed frequent fire return intervals, ranging from 3 to 38 years between fires of low- to mixed- severity (Agee 1993, Hessburg and Agee 2003). In forests with low- and mixed- fire severity regimes, fire plays critical roles in maintaining soil productivity, vegetative diversity, hydrological cycles, stand structure, and wildlife habitat (Kilgore 1973, Arno et al. 1995, Harvey et al. 1999, Harvey et al. 2000, Galley and Wilson 2001, Chong et al. 2003, Fitzgerald 2005, Graham and Jain 2005). The moderate temperatures and low humidity of these forests tend to foster slow decomposition rates, resulting in accumulation of biomass on the forest floor (Harvey 1994). Buildup of duff, litter, and fine woody debris supplies a consistent source of highly flammable fuels to carry fire after ignition, which has historically been provided primarily by lightning and indigenous sources (Barrett and Arno 1982, Pyne 1982). Fire releases organically bound nutrients to their inorganic forms, which increases availability for vegetation and microbial uptake (Covington and Sackett 1988, Harris

and Covington 1983). Low-severity fire has also been shown to increase soil moisture content and temperature, which can quicken microbial decomposition rates (Ryan and Covington 1986). Trees benefit from fire as early as germination, as surface fires prepare the seedbed by exposing bare mineral soil (Hartesveldt and Harvey 1967, Kilgore 1973). Recurrent fires also maintain a reduced canopy cover, providing gaps of sunlight for shade-intolerant seedlings. Many of the native shrubs and forbs are fire-dependent and sustain ungulates, birds, and other mammals as forage (Kay 1960, Kilgore 1973). As low-severity fires often release nutrients, forage also maintains a greater nutritional value (Lawrence and Biswell 1972). Given the slow decomposition rates in these arid forests, a hundred years of fire suppression has resulted in profuse quantities of both surface and canopy fuels, leading to uncharacteristic fire behavior and alterations to these ecological functions.

1.3 History and consequences of fire exclusion

Over the past century, European settlers have substantially altered the historic fire regime through land use alteration and fire suppression (Lunan and Habeck 1973, Agee 1993, Hessburg and Agee 2003, Naficy et al. 2010). Early colonizers caused a decrease in fire frequency through introduction of domestic grazers and eradication of indigenous populations, as natives across much of the west historically utilized burning for various objectives, including improvement of hunting visibility and promotion of forage species (Barrett and Arno 1982, Pyne 1982, van Wagendonk 2007). Institutional fire suppression policy was not initiated until the late 1800's with the formation of the first National Parks (Rothman 2007). Within all federal agencies, including the newly founded United States Forest Service (USFS), suppression remained the only acceptable response to both human- and natural- caused fire for the next several decades (Stephens and Ruth 2005). It wasn't until the late 1960's that the USFS and National Park

Service (NPS) finally conceded that fire had value as an ecological process and began allowing select fires to burn (van Wagtendonk 2007). Allowance of naturally caused burns and utilization of prescribed burns has gradually gained acceptance over the past 45 years, but continues to remain a contentious management option.

Fire-dependent forests with altered disturbance regimes have been shown to have decreased biodiversity, plant vigor, nutrient availability, as well as increased susceptibility to insects and disease, old-tree mortality, and seasonal drought (Covington et al. 1997, Keane et al. 2002). In the absence of disturbance, density has increased dramatically in many western forests, with reconstruction studies in the Southwest indicating some sites have as many as 56 times the amount of pre-settlement trees per acre (Covington et al. 1997). The vast majority of the additional trees are small, and in mixed-species forests, tend to be late seral, shade-tolerant species. This excess of trees populating the lower strata of the canopy provide ample ladder fuels to transfer surface fires into the canopy, and subsequently carry extensive crown fires through an uninterrupted canopy layer (Steele 1994, Keane et al. 2002, Pollet and Omi 2002, Graham et al. 2004, Graham and Jain 2005).

Buildup of excess of fuel increases both the behavior and severity of fire, where behavior describes the physical attributes of the fire (i.e. flame length, etc.) and severity refers to the effects of the fire (i.e. soil heating, tree mortality, etc.) (Keeley 2009). The damaging consequences of high-severity fires in ecosystems with low-severity fire regimes are well documented (Campbell et al. 1977, Cromack et al. 2000, Savage and Mast 2005, Dore et al. 2010, Roccaforte et al. 2012). High-intensity crown fires often result in extensive mortality of mature trees, and 100% mortality rates are not uncommon. The extreme temperatures reached during high-intensity fires generate severe consequences for soil, such as high rates of microbial

mortality, increased soil temperatures, nutrient leaching, and erosion from runoff (Neary et al. 1999, Certini 2005, MacDonald and Huffman 2004, Goforth et al. 2005). For instance, in a case study of a northern Arizona ponderosa pine wildfire, severely burned areas had 7 times more large tree mortality, 8 times more runoff, and 2 to 7.5 more Ca, Mg, and K leaching than the unburned control (Campbell et al. 1977).

1.4 Fuel reduction treatments

Manipulation of forest fuels, either through mechanical methods or prescribed fire is needed before wildfire can return into these systems without damaging consequences. The USFS currently spends more than half its budget on suppressing wildfires and a significant portion of management activities focus on reducing fuels to minimize future wildfire severity (USDA Forest Service 2015, National Interagency Fire Center 2016b). It is estimated that more than 66 million acres of the forested landscape in the western United States could benefit from fuel reduction, and in 2003 President Bush signed the Healthy Forests Restoration Act, recognizing the inevitability of future fires in various western ecosystems and necessitating prioritization of fuel reduction treatments in fire-prone forests (United States House of Representatives 2002). The law includes mandates for thinning overstocked stands, creating shaded fuel breaks, improving wildland firefighting techniques, and establishing community wildfire protection plans in the wildland urban interface (United States House of Representatives 2002). Fuel treatments attempt to decrease intensity and severity of wildfires when they reach treated areas, thereby reinstating healthy and resilient characteristics of fire-prone ecosystems by altering forest fuel structures and amounts (Agee and Skinner 2005, Finney and Cohen 2003, Ruth and Stephens 2005, Reinhardt et al. 2008).

Forest fuels are comprised of a vertical succession of strata: ground fuels, surface fuels, ladder fuels, and canopy fuels (Graham et al. 2004, Peterson et al. 2005). Ground fuels are subsurface materials, including duff, roots, and buried woody debris. Surface fuels consist of woody debris, herbaceous vegetation, and small shrubs, while ladder fuels refer to the layers of woody vegetation, such as large shrubs, saplings, and small trees, ascending towards the canopy. To be successful, any fuel reduction treatment should consider multiple aspects of the fuel profile. Surface fuels should be reduced to decrease the flame length of potential surface fires, which will lessen the chance of fire escalation into ladder fuels, while ladder fuels should be removed to minimize crown fire initiation. Decreasing overall stand density reduces the possibility of the canopy carrying an active crown fire after torching. Thus, to successfully restore resiliency in low-intensity, high frequency fire regimes, fuel reduction treatments should focus on: (1) reducing surface fuels, (2) increasing canopy base height by removing ladder fuels, (3) reducing canopy bulk density and continuity and (4) retaining large, fire-resistant species (Van Wagner 1977, Agee 1996, Graham et al. 1999, Scott and Reinhardt 2001, Cruz et al. 2002, Graham et al. 2004, Agee and Skinner 2005).

A frequently used method for reducing density in dry, mixed-conifer forests begins with thinning. In any fuel reduction thinning treatment, it is beneficial to thin small diameter trees that will carry a surface fire into the canopy. A low thinning removes smaller trees, best addressing the third and fourth objectives mentioned above: reducing canopy bulk density and retaining large trees (Agee and Skinner 2005, Graham et al. 1999). Thinning prescriptions also tend to retain shade-intolerant species with fire-resistant traits such as pine (*Pinus* spp.) and larch (*Larix* spp.) over shade-tolerant species, such as true firs (*Abies* spp.) and Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco). In the absence of fire, closed-canopy conditions favor shade-

tolerance, and the density of these species has increased substantially relative to shade-intolerant species, which tend to have more fire-resistant characteristics.

Thinning almost always results in a large quantity of non-merchantable harvested materials, as small trees and saplings are not large enough for the majority of commercial wood products. This non-commercial biomass results in large amounts of slash that generates additional surface fuels, known as activity fuels, to the surface fuel load unless subsequently treated or removed (Weatherspoon and Skinner 1995, Stephens 1998). Thus, an additional step in many fuel reductions entails treatment of activity fuels. Common applications are piling and burning, mechanical compaction, removal for biomass utilization, broadcast burning, or combinations and variations on these approaches (Graham et al. 2004). Mechanical treatments such as chipping and mulching aim to manipulate fire behavior by rearranging fuel loads instead of reducing them. Pile burning or removal of slash both assemble activity fuels for treatment, and can be very effective at reducing fuel loads. However, the concentration of heat from pile burning can detrimentally affect soil nutrients and microbial activity (Esquilin et al. 2007, Johnson et al. 2011). Similarly, removal of woody material from the forest floor can reduce availability of nutrients from decomposition of materials (Harvey et al. 1976, Bengtsson et al. 1998, Hacker 2005). While these options can be effective at reducing surface fuels, they lack the ecological benefits of broadcast burning, and in some cases, can have negative impacts on ecosystem functions.

Broadcast burning can be used alone or in conjunction with other fuel reduction strategies and provides numerous benefits. Burning consumes surface fuels, kills fire-intolerant regeneration, revitalizes understory vegetation, increases crown base height, and stimulates nutrient cycling (Ffolliot et al. 1977, Sackett 1980, Walstad et al. 1990, Peterson et al. 2005).

However, there are many additional considerations when applying prescribed fire, including weather conditions, proximity to human infrastructure, and safety. Additionally, sites often require preparation before fuel loadings are nominal enough to facilitate low-severity burns, especially when there are heavy accumulations of ladder and ground fuels.

1.5 Carbon storage in fire-frequent forests

One inadvertent benefit of the suspension of natural disturbance regimes and increases in forest biomass has been a net gain in carbon storage throughout many of the dry forest types in the West, which has helped to offset increases in anthropogenic fossil fuel emissions (Birdsey et al. 1992, Sohngen and Haynes 1997, Houghten et al. 2000, Hurtt et al. 2002). Recent attention has focused on the ability of forests to continue to offset emissions as atmospheric carbon levels continue to rise (Birdsey et al. 1993, Sohngen and Haynes 1997, Hudiburg et al. 2009). In fire-prone ecosystems, however, longer and drier fire seasons and a century of fuel accumulation are contributing to increasing risk of carbon offset reversal due to the occurrence of high-severity forest fires (Wiedinmyer and Neff 2007). Forest managers are now faced with the task of maximizing carbon storage while minimizing risk of offset reversal. It remains unclear as to which fuel reduction strategies, if any, best balance these two objectives. Intuitively, the objectives are conflicting, as fuel reduction treatments are explicitly removing biomass from ecosystems. However, there is potential for mutually beneficial treatment outcomes (Galik and Jackson 2009). For example, studies have suggested that the large trees on the landscape prior to European settlement stored more carbon than the abundance of smaller trees that currently exist (Fellows and Goulden 2008). Increasing the amount of biomass stored in large, fire-resistant trees is an objective that can provide both carbon storage and fire resilient stand structures.

The effects of fuel reduction treatments on aboveground biomass distributions is currently unknown. Better understanding of aboveground biomass distribution after fuel treatments can be used to inform models that attempt to quantify carbon emissions from potential wildfire with and without previous fuel treatments. While it is apparent that fuel reduction treatments can exponentially decrease carbon offset from wildfire (Hurteau et al. 2008, Hurteau and North 2009), it remains unclear whether the net emissions from wildfires in an untreated stand surpass those of a fuel reduction treatment and wildfire combined (Reinhardt and Holsinger 2010). Better understanding of the tradeoffs and consequences of treating a stand as opposed to simply maximizing carbon stocks will inform policies such as the California Climate Action Registry (CCAR), which are based on the assumption that wildfires can continue to be excluded from fire-frequent forests, and which penalize managers for thinning operations (California Climate Action Registry 2007, Hurteau et al. 2008).

Over the past two decades, several studies have attempted to characterize modifications of aboveground biomass from fuel reduction treatments. Different combinations of thinning and burning are likely to remove different forms of biomass. For example, while thinning explicitly removes overstory biomass, burning without thinning may consume more of the understory and leave the overstory virtually intact, or alternatively, increase coarse woody biomass from overstory mortality. While only a few studies have examined treatment effects on total aboveground biomass, many have characterized treatment effect on specific components of the biomass distribution, i.e. responses of overstory, understory, and woody biomass. This review will survey responses of total aboveground biomass and individual biomass components to common types of fuel treatments, as well as examine predicted effects on the overall distribution of aboveground biomass.

1.6 Responses of biomass to fuel treatments

1.6.1 Total aboveground biomass

A common objective of fuel treatments is to reduce combustible forest biomass. With few exceptions, overall stand biomass has been shown to decrease, at least initially, as a result of fuel treatments across various fire-frequent forest types, including Sierra Nevada mixed-conifer, mixed and pure ponderosa pine in both the eastside Cascades and southwestern United States, and all 12 Fire and Fire Surrogate study sites (Boerner et al. 2008, Finkral and Evans 2008, Mitchell et al. 2009, North et al. 2009, Stephens et al. 2009, Sorensen et al. 2011) (Table 1). However, the duration of this reduction in biomass is still unclear. For example, in a meta-analysis of a national network of 12 Fire and Fire Surrogates studies (FFS) initial measurements exhibited declines in total ecosystem carbon immediately post-treatment (within 1 year) for thinning, burning, and thinning and burning treatments relative to the control; however, subsequent re-measurements often exhibited similar total average carbon stores as the controls (Boerner et al. 2008). In a modeled scenario in ponderosa pine stands of the eastern Cascades, Mitchell et al. (2009) predicted that decreases in biomass would persist for at least 10 years after treatment when fire was applied, either with or without a preceding thinning. However, thinning without burning would result in greater levels of biomass relative to the control. As there are currently few studies with biomass re-measurements extending past three years after treatment, duration of total ecosystem biomass reductions after fuel treatments is still unknown.

Table 1. Responses of total aboveground biomass to fuel treatments relative to the control (* indicates that the responses were modeled).

Total Aboveground Biomass				
Location	Treatment	Time since treatment	Response	Reference
East Cascades ponderosa pine	burn	1 year	increase	Boerner et al. 2008
Montana ponderosa pine	burn	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	burn	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	burn	3 years	decrease	Stephens et al. 2009
East Cascades ponderosa pine	burn	10 years *	decrease	Mitchell et al. 2009
Arizona ponderosa pine	thin	immediate	decrease	Sorensen et al. 2011
Arizona ponderosa pine	thin	immediate	decrease	Finkral and Evans 2008
Sierra mixed-conifer	thin	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	thin	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	3 years	same	North et al. 2009
Sierra mixed-conifer	thin	3 years	decrease	Stephens et al. 2009
East Cascades ponderosa pine	thin	10 years *	increase	Mitchell et al. 2009
Sierra mixed-conifer	thin/burn	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	thin/burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	3 years	decrease	Stephens et al. 2009
East Cascades ponderosa pine	thin/burn	10 years *	decrease	Mitchell et al. 2009
East Cascades ponderosa pine	thin/burn	10 years *	decrease	Mitchell et al. 2009

1.6.2 Live tree biomass

Live trees are the largest store of biomass in the majority of western forests, especially given the current high stand densities of many dry, fire-frequent forest types that were historically less dense (Boerner et al. 2008). As the majority of fuel treatments involve thinning,

these treatments universally result in an immediate decrease in live tree biomass (Boerner et al. 2008, Campbell et al. 2009, Mitchell et al. 2009, North et al. 2009, Stephens et al. 2009, Sorensen et al. 2011) (Table 2). However, over time the increases in residual tree growth rates replaces harvested biomass, and stand-level biomass storage can approach that of pre-harvest conditions (Campbell et al. 2009). Although there are few studies with long enough durations to adequately follow tree growth post-treatment, historical reconstructions indicate that pre-settlement forests composed of fewer large-diameter trees stored more live-tree carbon than the numerous small diameter trees that are currently on the landscape (Fellows and Goulden 2008, Hurteau and North 2009). This suggests that the release from competition and increase in growing space from thinning may eventually result in greater amounts of biomass stored in the overstory relative to an untreated stand.

The addition of prescribed fire after a thinning may or may not contribute to further decreases in live tree biomass as a result of post-fire mortality (Boerner et al. 2008, North et al. 2009, Stephens et al. 2009) (Table 2). For example, North et al. (2009) witnessed successively greater reductions in residual live-tree biomass with escalations in treatment intensity, where broadcast burning after both understory and overstory thinning treatments further decreased post-treatment (1 year) carbon stores in live trees by 6 and 9 percent, respectively, from fire-caused mortality. Furthermore, prescribed burning may inhibit future biomass accumulation of residual trees. In a study examining the effect of various combinations of thinning and burning on rates of biomass accumulation eight years after treatment, Hurteau and North (2010) observed that the greatest rate of live-tree carbon accumulation, as well as overall live-tree carbon storage, was accomplished using an understory thinning without burning. The understory thin/burn and burn-only units experienced almost half of the carbon gains in large trees over the same duration of

time, possibly attributed to greater occurrences of post-treatment mortality in the burning treatments. Within the same study however, carbon gains from the overstory thin and burn treatment was the same as the overstory thin without burning. Similarly, in the FFS meta-analysis, there was no difference between residual vegetative biomass in thinned versus thinned and burned treatments (Boerner et al. 2008).

While thinning before burning creates some consistency in pre-burn stand conditions, burning without prior thinning allows for a wide range of pre-burn conditions across study sites, resulting in highly inconsistent treatment responses. For instance, burning without thinning appears to have mixed effects on residual live-tree biomass (Boerner et al. 2008, North et al. 2009, Stephens et al. 2009) (Table 2). While several studies, including the FFS meta-analysis, found no difference in live-tree carbon in burn-only units relative to the control (Boerner et al. 2008, Stephens et al. 2009), others witnessed decreases in carbon ranging from 6.8-18% (Boerner et al. 2008, North et al. 2009). These discrepancies are likely due to high variability in burning conditions and fire behavior during treatments, which produce variable fire effects on live trees.

Table 2. Responses of live tree biomass to fuel reduction treatments relative to the control.

Live Tree Biomass				
Location	Treatment	Time since treatment	Response	Reference
Sierra mixed-conifer	burn	3 years	decrease	Boerner et al. 2008
Sierra mixed-conifer	burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	burn	3 years	same	Stephens et al. 2009
Arizona ponderosa pine	thin	immediate	decrease	Sorensen et al. 2011
Arizona ponderosa pine	thin	immediate	decrease	Finkral and Evans 2008
Sierra mixed-conifer	thin	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	3 years	decrease	Campbell et al. 2009
Sierra mixed-conifer	thin	3 years	decrease	Stephens et al. 2009
Sierra mixed-conifer	thin	16 years	decrease	Campbell et al. 2009
Sierra mixed-conifer	thin/burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	3 years	decrease	Stephens et al. 2009

1.6.3 Understory biomass

As fuel treatments in fire-frequent forest types generally involve canopy removal, whether by mechanical removal or fire-caused mortality from burning, understory production tends to increase, both from better access to sunlight and reduced competition for belowground resources (Connell and Smith 1970, Campbell et al. 2009). In a thinning study in a Sierra Nevada mixed-conifer forest, understory biomass increased almost threefold relative to the control within the three years following treatment (Campbell et al. 2009). Understory vegetation has been shown to respond positively to both increases in direct sunlight as well as overstory quadratic mean diameter (QMD) (Connell and Smith 1970). Fewer, larger trees allow a greater portion of light to penetrate the canopy, as foliage tends to be less dense (Naumberg and DeWald 1999). As the overstory canopy recovers from the initial thinning and canopy cover increases,

understory plants experience shading and production decreases (Campbell et al. 2009).

Therefore, understory responses are highly dependent on thinning intensity, as well as the development of the residual overstory over time.

Many understory species in fire-dependent forests either benefit from or require fire for reproduction and germination (Brown and Kapler 2000). Prescribed fire may also decrease litter depth, which can reduce the effort needed to germinate, and increase nutrients available for uptake, increasing growth rates. However, observed increases in production after thinning and burning treatments may be primarily attributed to overstory removal (Ryan and Covington 1986, Gaines et al. 1958, Ffolliott et al. 1977, Sackett 1980). For instance, Kane et al. (2010) evaluated understory plant cover among five different fuel treatment types: mastication, mastication followed by tilling, mastication followed by prescribed fire, hand removal, and control. All treatments resulted in at least a twofold increase in understory plant cover, but that increase did not appear to differ among applications, indicating that understory vegetation was primarily responding to increased light availability and growing space.

Nonetheless, many studies evaluating prescribed burning alone describe an overall increase of understory biomass, demonstrating that fire does provide benefits to understory production (Ffolliott et al. 1977, Andariese and Covington 1986, Vose and White 1991, Huisinga et al. 2005). These responses to fire appear to vary greatly depending on species, plant size and vigor, fire behavior, seasonality of the broadcast burn, and climate conditions. For instance, Oswald and Covington (1984) simultaneously observed an increase in herbage production and decrease in forage production after a prescribed burn relative to the unburned treatment. Additionally, understory biomass responses may differ among overstory stand structures. Harris and Covington (1983) found that understory biomass doubled after a prescribed burn relative to

unburned units for stands primarily comprised of trees less than 11 inches dbh. However, there was no significant difference between burned and unburned treatments for stands composed of primarily larger (> 11 in. dbh) trees. As with thinning, responses of understory production to burn-only treatments may change over time. For example, Andarieese and Covington (1986) noted that increases in production were seemingly delayed for the first two years after treatment.

1.6.4 *Forest floor biomass*

Forest floor biomass plays a critical role in propagation of fire, as well as nutrient cycling, wildlife habitat and hydrology (Harmon et al. 1986, Brown et al. 2003). Forest floor biomass in this context refers to the duff, litter, fine woody debris (FWD) and coarse woody debris (CWD) that constitute ground and surface fuels. As thinning tends to result in slash from logging operations, forest floor biomass might be expected to increase after harvesting. However, this is not always the case. Our review indicates that CWD tended to either decrease or remain the same relative to controls (Stephens and Moghaddas 2005a, North et al. 2009) (Table 3). Some studies observed increases in litter and FWD, ranging from 18-28%, within the first few years after harvest (e.g. Stephens and Moghaddas 2005b, North et al. 2009), while others witnessed decreases, ranging from 10-24% (e.g. North et al. 2009, Sorensen et al. 2011). Several studies detected no change at all (Boerner et al. 2008, Finkral and Evans 2008, Stephens et al. 2009). A potential consideration when evaluating the effects of fuel treatments on litter and FWD is the residual overstory species composition. For instance, in a mixed-conifer forest in the Sierra Nevada, greater white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) abundances were associated with larger quantities of fine woody debris, while higher densities of pine were associated with relatively less fine woody debris and more litter (Lydersen et al. 2015). The method in which a stand is logged may also influence the distribution of woody

debris after thinning. CWD can be generated from logging damage to limbs and small diameter trees, and may increase with time since harvest, as damaged trees die and eventually fall. In the Fire and Fire Surrogate study, many of the thinning sites had similar amounts of CWD pre- and post- harvesting, despite differences in harvesting methods. However, at the Washington site, which used helicopter logging, CWD increased by 100-150% (Boerner et al. 2008).

When fire is added to the treatment, studies almost unanimously observe a decrease in duff, litter, and FWD from consumption during the burn (Stephens and Moghaddas 2005b, Boerner et al. 2008, North et al. 2009, Stephens et al. 2009) (Table 3). This is to be expected, as fine, relatively dry, dead biomass materials are the primary source of fuels carrying surface and ground fires. Combinations of thinning and burning also tended to result in significant decreases of CWD (North et al. 2009, Stephens and Moghaddas 2005a, Stephens and Moghaddas 2005b). One critical consideration regarding woody debris when evaluating fuel treatments is the sensitivity of woody biomass responses to fire intensity. For example, CWD is typically categorized by levels of decay; all other factors held equal, biomass that is more decayed ignites more easily than sound woody material (Harmon et al. 1986). In one study of CWD responses to fuel treatments, the two sound decay classes showed no decrease in overall volume in the thinned and burned units, while the volume of logs with greater decay significantly decreased. Similarly, the amount of residual large CWD was similar between the controls and treatments that included fire, while the amount of smaller CWD decreased in volume under the same treatments (Stephens and Moghaddas 2005a).

Burning without thinning tends to decrease forest floor biomass (Knapp et al. 2005, Stephens and Moghaddas 2005a, Stephens and Moghaddas 2005b, Boerner et al. 2008, North et al. 2009, Stephens et al. 2009) (Table 3). One consideration for managers is the season of burn,

as in the continental, Mediterranean climates summer and fall burns tend to be more consistent with historical burning patterns, but fuel moisture content tends to be higher in the spring, which can lessen fire severity (Alexander 1982, Martin and Sapsis 1992, Caprio and Swetnam 1995). For instance, in a mixed-conifer forest in the southern Sierra Nevada, spring burns consumed 67% of the total dead and downed woody biomass, while the fall burns consumed 88% of total downed woody biomass (Knapp et al. 2005). Development of woody biomass composition over a temporal scale is also an important consideration. For example, a burn with higher severity may initially consume greater amounts of coarse woody debris relative to a lower severity burn but may also result in greater live-tree mortality rates, which may eventually yield greater quantities of coarse woody debris relative to a lower severity burn. Thus, managers administering burning treatments need to consider how climate conditions will affect burning severity, and how fire behavior will alter woody biomass distributions over time.

Table 3. Response of forest floor biomass: duff, litter, fine woody debris (FWD), and coarse woody debris (CWD) to fuel reduction treatments.

Forest floor Biomass					
Forest type	treatment	Response variable	Time	Response	Reference
Sierra mixed-conifer	burn	forest floor	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	burn	forest floor	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	burn	CWD	3 years	decrease	North et al. 2009
Sierra mixed-conifer	burn	fine woody debris	3 years	decrease	North et al. 2009
Sierra mixed-conifer	burn	litter	3 years	decrease	North et al. 2009
Sierra mixed-conifer	burn	litter, duff, surface wood	3 years	decrease	Stephens et al. 2009
Sierra mixed-conifer	thin	CWD	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	CWD	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	fine woody debris	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin	fine woody debris	3 years	increase	North et al. 2009
Sierra mixed-conifer	thin	litter	3 years	increase	North et al. 2009
Sierra mixed-conifer	thin	litter	3 years	increase	North et al. 2009
Sierra mixed-conifer	thin	litter, duff, surface wood	3 years	same	Stephens et al. 2009
Arizona ponderosa pine	thin	surface fuels	immediate	decrease	Sorensen et al. 2011
Arizona ponderosa pine	thin	surface fuels	immediate	same	Finkral and Evans 2008
Sierra mixed-conifer	thin/burn	forest floor	1 year	decrease	Boerner et al. 2008
Sierra mixed-conifer	thin/burn	CWD	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	CWD	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	fine woody debris	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	fine woody debris	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	litter	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	litter	3 years	decrease	North et al. 2009
Sierra mixed-conifer	thin/burn	litter, duff, surface wood	3 years	decrease	Stephens et al. 2009

1.6.5 Standing dead tree biomass

Our review indicates that the volume of snags either increased or stayed the same after thinning relative to controls, where mortality was a result of logging damage during harvesting (Stephens and Moghaddas 2005a, Boerner et al. 2008, North et al. 2009). Snags showed mixed responses to thinning and burning treatments, and were seemingly dependent on burn intensity (Stephens and Moghaddas 2005a, North et al. 2009). In one case, density of snags increased

while stand-level biomass in snags remained the same, indicating that smaller trees may have experienced mortality in response to the burn, while some of the older, larger snags fell and transferred to coarse woody debris (Stephens and Moghaddas 2005a). As with thinning and burning, burn-only treatments typically resulted in increased or similar snag biomass relative to the controls, (Stephens and Moghaddas 2005a, North et al. 2009). These responses are all highly dependent on fire severity, which is often highly variable, both within and among prescribed burns. Furthermore, the effect of fire on a snag can depend greatly on the decay status, species, and size of the snag, as well as site quality and wood density of the snag.

1.6.6 Aboveground biomass distribution

Although many studies have focused on the effects of fuel treatments on individual biomass components, very few have examined treatment effects on the distribution of aboveground biomass. One comparison of thinning and burning combinations in a Sierra Nevada mixed-conifer forest observed a negative relationship between treatment intensity and the relative proportion of carbon stored in live trees, including roots (North et al. 2009). The removal of live-tree carbon markedly altered the relative distribution of remaining carbon, proportionally increasing biomass in snags, soil, and coarse woody debris, while decreasing storage in roots and surface fuels. This effect was magnified with the addition of fire, and for the greatest intensity treatment (overstory thin and burn) the percentage of carbon in soils surpassed the proportion in live trees (North et al. 2009). Although that is only a single study, it illustrates a short-term (3 years) negative association between treatment intensity and the proportion of ecosystem biomass stored in the overstory, illustrating how treatments can substantially alter the distribution of aboveground biomass.

Inferences can be made regarding the relative proportion of biomass by summarizing responses of each ecosystem component to treatments. Thin-only treatments tended to observe decreases in overstory biomass, increases in understory biomass, decreases or no change in CWD biomass, increases or no change in snag biomass, and mixed results in the fine forest floor biomass (Stephens and Moghaddas 2005a, Stephens and Moghaddas 2005b, Boerner et al. 2008, Finkral and Evans 2008, Campbell et al. 2009, North et al. 2009, Stephens et al. 2009, Kane et al. 2010, Sorensen et al. 2011). Within zero to three years since treatments, treatments with thinning and burning showed decreases in overstory biomass, surface biomass, CWD biomass, increases in understory biomass, and mixed results for biomass in snags (Stephens and Moghaddas 2005a, Stephens and Moghaddas 2005b, Boerner et al. 2008, North et al. 2009, Stephens et al. 2009, Kane et al. 2010). Finally, burning without thinning treatments typically resulted in decreases in forest floor biomass, CWD biomass, decreases or no change in overstory biomass, and either increases or no change in both snag biomass and understory biomass 0 to 3 years after treatment (Oswald and Covington 1984, Vose and White 1991, Knapp et al. 2005, Stephens and Moghaddas 2005a, Stephens and Moghaddas 2005b, Boerner et al. 2008, North et al. 2009, Stephens et al. 2009).

Existing research on the effects of fuel treatments on aboveground biomass has been limited to observations of treatment responses within only a few years since implementation, and it remains unclear as to how long treatment effects persist, or how the biomass profile develops over time. It is apparent, however, that the majority of fire-frequent forest ecosystems require regimes of recurring fuel reduction treatments to be effective at perpetuating low-severity fire (Reinhardt et al. 2008). A common uncertainty is how often re-entries should occur, or which treatments might prolong the need for repeated management (Jain et al. 2012). Analysis of long-

term data are needed to evaluate how long these distributions of biomass develop temporally, and which treatments might maximize fuel treatment effectiveness for the longest period of time while balancing the need to manage carbon storage.

1.7 Conclusion

From this review, it is clear that there are at least two knowledge gaps in the current research regarding the effects of commonly used fuel reduction treatment on biomass accumulation and distribution in the fire-prone forests of the American West. The first gap is treatment longevity. While several studies have measured post-treatment biomass up to 3 years after harvesting and/or burning, there have yet to be any studies on the duration of these effects over the long-term. Secondly, there have been no studies examining the effects of fuel treatments on biomass in northern Rocky Mountain ecosystems. The northern Rockies are unique in that many fire-frequent forests here have longer fire return intervals relative to other regions (Arno et al. 1995, Arno et al. 1997). As a result, they historically tended to have higher densities and less uniformly distributed spacing relative to ponderosa pine forests of the Southwest or the Pacific Northwest (Clyatt et al. 2016). Differences in historic disturbance regimes and stand densities are likely to produce unique responses to fuel reduction treatments relative to other regions.

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SECTION 2:

23-year stand dynamics and tree growth after restoration treatments in ponderosa pine
forests of the Northern Rocky Mountains

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Abstract

Many low-elevation, dry forests types of the western United States historically experienced frequent, low-severity forest fires. European settlement and fire suppression policies resulted in over a century of fire exclusion, greatly altering forest structure and increasing severity of fire behavior. There is considerable interest in restoring fire-resilient characteristics to these forests through restoration treatments. Yet, few studies have quantified long-term responses of tree growth and stand dynamics to commonly applied restoration treatments. Our study examined 23-year growth responses to restoration treatments in a ponderosa pine dominated forest in western Montana in two separate studies with different silvicultural prescriptions consisting of cutting (shelterwood and thinning), followed by prescribed burning. We show that stand densities were still lower in all treated stands relative to the control, while stand-level basal

area increment was the same across all treatments and tree-level basal area increment tended to be greater in the treated stands relative to the control. Growth efficiency tended to be the same across treatments while leaf area index was either the same or lower than the control. Treatments that included burning appeared to best address restoration objectives by promoting growth of large trees and successfully recruiting a new cohort of pine regeneration in both the thinning and shelterwood installations.

2.1 Introduction

In the western United States, low-elevation, dry forest types are common across the landscape. These forests are often characterized by a history of low- to mixed- severity forest fires, which occurred on intervals ranging from 3 to 50 years (Hessburg and Agee 2003, Fitzgerald 2005). Over the past century, European settlers have substantially altered the historic fire regime through land use changes and fire suppression policies (Pyne 1982, Agee 1993, Hessburg and Agee 2003, Stephens and Ruth 2005, van Wagtendonk 2007, Naficy et al. 2010). Consequences of these suppression policies have become apparent over the past few decades, as wildfires have increased drastically in severity, intensity, and magnitude in many of these forests (Keane et al. 2002, North et al. 2015). For example, in 2015 wildfires burned more than 10 million acres in the U.S., primarily in Alaska and the western states (NIFC 2015). Since the National Interagency Fire Center began collecting data in 1960, only 6 other years have surpassed 8 million burned acres, all occurring since 2004 (NIFC 2015). With changes in climate predicted to cause increases in fire season longevity and extreme weather conditions, there is considerable interest in restoring historic fire regimes to the low- elevation, dry forest types of the west (Flannigan et al. 2006, Westerling et al. 2006, Flannigan et al. 2013, Van Mantgem et al. 2013 Jolley et al. 2015).

A primary consequence of fire suppression has been the alteration of forest structure and composition. In many of the fire-frequent forests experiencing fire exclusion, overstory composition has experienced shifts towards a greater abundance of mid-story small trees and saplings. For example, between 1952 and 1992, the volume of trees smaller than 17 inches in diameter increased by 52%, and in 1992 accounted for two-thirds of the trees on much of the western landscape (Powell et al. 1993). In addition to reducing individual tree vigor from overcrowding, this development of regeneration has created a matrix of ladder fuels that allow surface fires to transition to crown fires (Keane et al. 2002, Graham et al. 2004). Moreover, shifts in species composition from shade-intolerant species to shade-tolerant species have increased the number of fire intolerant species on the landscape and contributed to contiguity of ladder fuels, as shade-tolerant species are more likely to have crown base heights in proximity to the forest floor.

In the mixed- and pure- ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests of the Northern Rocky Mountains, frequent fire historically promoted ponderosa pine over concurrent, shade-tolerant species such as Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.)) (Jain et al. 2012). Historic mean fire return intervals in this region varied from 3 years to upwards of 50 years (Arno et al. 1995, Arno et al. 1997, Hessburg and Agee 2003, Heyerdahl et al. 2008). In many of these forests, fire has been excluded from the landscape since the early 1900's (Agee 1993, Keane et al. 2002, Hessburg and Agee 2003). In the absence of fire-induced regeneration mortality, stem densities of Douglas-fir have increased, altering species composition, and increasing the likelihood of high-severity fire (Dodge 1972, Keane et al. 2002, Keeling et al. 2006).

There have been very few opportunities to quantify long-term empirical responses to restoration treatments in mixed- and pure- ponderosa pine stands. The majority of studies have been limited to quantifying treatment responses within a decade or less of implementation. Additionally, much of the research conducted on restoration treatments in ponderosa pine comes from the Southwest, or other regions. The Northern Rockies are unique in that many fire-frequent forests here have longer fire return intervals relative to other regions (Arno et al. 1995, Arno et al. 1997). As a result, they previously tended to have higher densities and less uniformly distributed spacing relative to ponderosa pine forests in the Southwest or the Pacific Northwest (Clyatt et al. 2016). Differences in historic disturbance regimes and stand densities are likely to create unique responses to restoration treatments relative to other regions (e.g. Keeling et al. 2006). More information is needed regarding region-specific vegetation responses to restoration treatments over longer durations of time.

The Lick Creek Demonstration/Research Forest includes two of the longest running studies addressing potential implications of restoration treatments in ponderosa pine forests of the Northern Rockies. An experiment incorporating two different experimental silvicultural strategies was initiated in 1991. Within each silvicultural strategy, two different broadcast burning prescriptions were applied to determine potential additive benefits of prescribed fire. A retention shelterwood was applied to an 80 year old stand situated at the base of the drainage and a commercial thinning was applied to a second stand, located upslope of the drainage and approximately 70 years of age, with primarily pole-sized trees (Smith et al. 1999). Together, these studies at Lick Creek provide a unique opportunity to describe forest stand structure 23 years after treatment and determine differences in vegetation response to different burning prescriptions. We evaluated each of the two selected silvicultural installations individually, as

differences in each installation's management history and site conditions inhibit direct comparison between them. Our objectives were to (1) determine stand structure and composition 23 years after treatment, (2) evaluate trends in growth since time of treatment, and (3) characterize regeneration patterns and composition.

2.2 Methods

2.2.1 Study site

The Lick Creek Demonstration/Research Forest (hereafter: Lick Creek) is located on the Darby Ranger District of the Bitterroot National Forest in southwestern Montana (46°5'N, 114°15'W) (Figure 1a). The site is semi-arid, with an estimated average annual temperature of 45 °F and precipitation of 16 inches, with about 30% of this annual precipitation falling as snow (Gruell et al. 1982, DeLuca and Zouhar 2000). Elevations within the demonstration forest range from approximately 4300 to 5000 feet, with slopes primarily ranging from 0 to 30 percent (Menakis 1994). Soils are relatively shallow or moderately deep, and are classified as Elkner Gravelly Loam, coarse-loamy, mixed, frigid Typic Cryochrepts, with highly weathered granite parent material (DeLuca and Zouhar 2000).

Overstory vegetation consists principally of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson var. *ponderosa* C. Lawson), with Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.)) in the understory, although grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*), and lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *latifolia* Engelm. ex S. Watson) are also intermittently present. Habitat types as classified by Pfister et al. (1997) within the drainage are Douglas-fir/snowberry (*Symphoricarpos albus* (L.) S.F. Blake) and Douglas-fir/pinegrass (*Calamagrostis rubescens*

Buckley.)) located on the southerly aspects, and Douglas-fir/dwarf huckleberry (*Vaccinium caespitosum* Michx.), blue huckleberry (*Vaccinium globulare* Douglas ex Torr.), twinflower (*Linnaea borealis* L. subsp. *americana* (Forbes) Hultén ex R.T. Clausen) and grand fir/twinflower on the northwest aspects (Menakis 1994).

The Lick Creek photoseries project was launched in the Lick Creek drainage of the Bitterroot National Forest immediately after the first large USFS timber harvest in ponderosa pine in the region occurred here (Gruell et al. 1982). The photos were taken at established photopoints throughout the Lick Creek drainage approximately every 10 years starting in 1909, allowing for a comprehensive depiction of forest stand dynamics over time (Figure 1c). Decades after the initial timber harvest, which took place from 1907 to 1911, portions of the drainage experienced various combinations of commercial thinnings and stand improvement cuttings, starting in 1952 and ending in 1980 (Menakis 1994). The effects of these additional cuttings were documented photographically along with untreated areas. Based on the photoseries and additional fire history reconstructions in the area, there is strong evidence that frequent, low-intensity fires or silvicultural surrogates are needed to maintain fire-resilient stand structures dominated by ponderosa pine (Gruell et al. 1982; Smith and Arno 1999).

In 1992, recommendations from the photoseries analysis motivated an experiment incorporating two different silvicultural strategies. Within each silvicultural strategy, two different broadcast burning prescriptions were applied to determine potential additive benefits of prescribed fire. Treatments were designed to promote stands with predominantly large diameter pines, and recruit younger pine age classes to maintain an uneven-aged stand structure through time (Carlson et al. 1993). Among the objectives of the experimental installations were to determine which treatments would effectively remove shade-tolerant species from the understory

and promote natural regeneration of pine, as well as examine the effects of fire on diameter and height growth of residual trees.

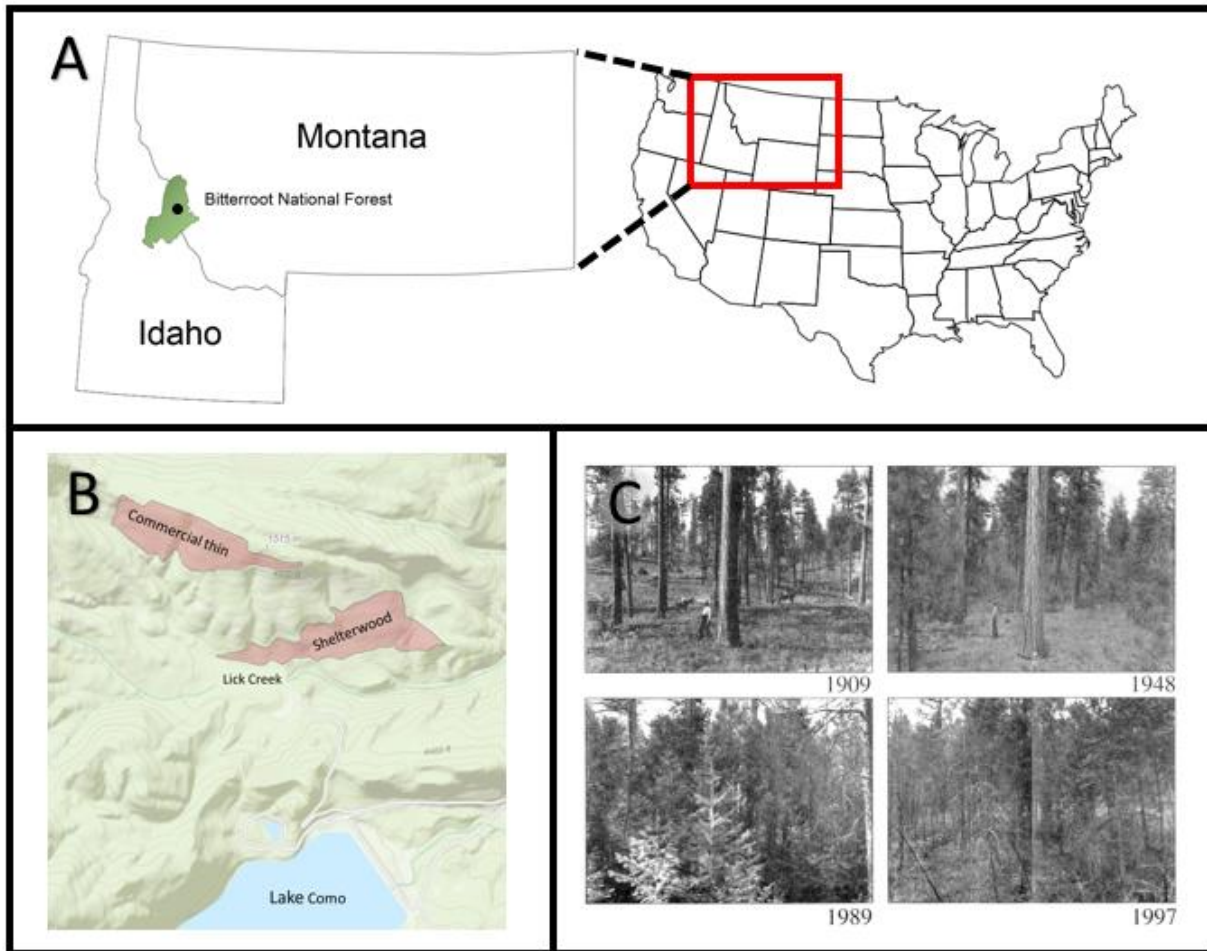


Figure 1. (a) The study site (black circle) is located on the Bitterroot National Forest in western Montana near the Idaho-Montana state border. (b) The two silvicultural installations are located between 4300 and 5000 ft. in elevation on south facing slopes in the Lick Creek drainage. The shelterwood is located downslope near Lick Creek, while the thinning is upslope, in proximity to the ridge. (c) The Lick Creek drainage is the location of the well-known photo series documenting forest succession after the first large USFS timber sale in ponderosa pine in 1907. Photos have been taken approximately every decade since 1909 at established photo points (Gruell et al. 1982).

2.2.2 Treatment description

The first installation is a commercial thinning located on the upslope of the Lick Creek drainage, with a south aspect and elevation of 4790 to 5050 feet (Figure 1b). This stand had been subjected to selection cutting administered from 1907 until 1911 by the USDA Forest Service, which had a goal of removing 70% of the ponderosa pine and all Douglas-fir greater than 10 inches in diameter (Menakis 1994). A commercial thinning and stand improvement cutting were completed in 1956, and in some portions of the stand, again in 1967 and 1980 (Menakis 1994).

By 1991, the stand had an average age of 70 years, with 85-100 ft² per acre of basal area (BA), approximately 170 trees per acre (TPA), and a 93% ponderosa pine species composition (Table 1). In 1991 the stand was divided into 12 units of 7 to 10 acres. Nine of the units were thinned in 1992 to a target residual basal area of 50 ft² acre⁻¹. Thinning targeted the removal of the smallest merchantable stems, and retained the largest and healthiest trees. The post-treatment stand had an average 61 ft² acre⁻¹ basal area and 112 TPA. In order to examine the effects of burning and burn seasonality, three units were burned in the fall of 1993, three units were burned in the spring of 1994, and three units were left unburned (Arno 1999). Three additional units were left unthinned and unburned, to serve as a control (Arno 1999).

The shelterwood is positioned towards the base of the drainage, with a primarily southerly aspect and elevations ranging from 4330 to 4560 feet (Figure 1b). This portion of the drainage was privately owned in the early 1900's, and was clearcut in 1906 and regenerated naturally. Besides a light stand improvement cutting in 1960, no other management was applied until the 1992 harvest; the stand was about 85 years old at the time of cutting (Gruell et al. 1982, Menakis 1994). Prior to harvesting, the stand supported 120 ft² acre⁻¹ BA, 240 TPA, and had a

72% ponderosa pine species composition (Table 1). The shelterwood cutting aimed to retain the largest and healthiest pines while reducing the density to 40 ft² acre⁻¹ BA in order to open the canopy for development of pine regeneration (Arno 1999). Post-treatment density averaged 52 ft² acre⁻¹ and 92 TPA (Smith et al. 1999). In addition to the overstory cutting, thinning was also applied to several dense pockets of smaller trees in three of the fuel reduction treatment units in order to normalize understory densities. Post-harvest burning treatments in the shelterwood consisted of a high consumption burn (lower duff was 16% moisture content) in three of the 12 units, a low consumption burn in three units (lower duff was 50% moisture content), and a no burn treatment in three cut units (Smith et al. 1999). However, high consumption did not occur in the dry duff units, as a rain shower near the end of the burns increased fuel moistures and reduced smoldering combustion. Three units were also left uncut and unburned to serve as a control.

Harvesting was conducted in 1992 in both the thinning and shelterwood units using chainsaw felling, followed by winch yarding to selected trails and skidding by a crawler tractor. Aside from the tree tops, which were cut at 6 inches diameter (outside bark), all other limbs were yarded to the landing with the bole and removed at the roadside landing (Arno 1999). Broadcast burning occurred in the spring and fall of 1993, as well as the spring of 1994 (Table 1). Treatment definitions for each of the two installations are as follows: fall burn in the thinning (FB); spring burn in the thinning (SB); dry burn in the shelterwood (DB); wet burn in the shelterwood (WB); the cut and unburned treatments are referred to as the No Burn in both the thinning and shelterwood (NB); and the control is identified as CO (Table 1).

Table 1. Treatment description of the 1992-1994 treatments at the Lick Creek Demonstration Forest for both the thinning and shelterwood studies.

Thinning							
Treatment	Abbreviation	Pre-treatment		Treatment objectives		1 year after treatment	
		TPA	BA	Cutting target (ft ² acre ⁻¹ BA)	Fuels treatment	TPA	BA
Thin and Fall Burn	FB	161	99	50	1993 Fall burn	96	64
Thin and Spring Burn	SB	142	84	50	1994 Spring burn	90	56
Thin and No Burn	NB	145	92	50	None	80	56
Control	CO	-	-	None	None	151	104

Shelterwood							
Treatment	Abbreviation	Pre-treatment		Treatment		1 year after treatment	
		TPA	BA	Cutting target (ft ² acre ⁻¹ BA)	Fuels treatment	TPA	BA
Cut and Dry Burn	DB	162	112	40	May 1993 high consumption burn	76	55
Cut and Wet Burn	WB	178	113	40	May 1993 low consumption burn	67	49
Cut and No Burn	NB	188	125	40	None	68	54
Control	CO	-	-	None	None	149	114

2.2.3 Data collection

Within each of the 12 units per installation is a systematic grid of 12 1/10th acre permanent circular plots. All trees (≥ 4 inches in diameter at breast height (dbh)) and saplings (<4 in. dbh and ≥ 4.5 ft. tall) were measured in 1991, 1993, 2005, and 2015 in each plot. Seedlings (<4.5 ft. tall) were tallied by species and height class in 1/100th acre nested subplots. Trees greater than 6 in. dbh were tagged in 1993, and in 1991, 1993, and 2005, species and

diameters were measured on all trees greater than breast height. Heights were also measured in 1991, 1993, and 2005 on 2 trees per plot in each of the following dbh classes: 0-4 in., 4-8 in., and > 8 in. All other heights were fitted via regression using parameters from height diameter relationships developed from the measured tree heights. In 2015, species, diameter, height, crown ratio, and crown base height were measured for all trees > 4 in. dbh.

Stem density (trees per acre; TPA), basal area (ft^2 per acre; BA), quadratic mean diameter (QMD), and stand density index (SDI, Reineke 1933) were all calculated by species using 2015 tree data. Only the tallest 100 trees per treatment were used to test differences in QMD of the most dominant overstory trees (QMD₁₀₀). Basal area and volume (ft^3) were calculated for 1991, 1993, 2005, and 2015. Volume was calculated using region- and species- specific equations developed by Flewelling and Raynes (1993) and administered by the Inland Northwest Growth & Yield Cooperative (INGY) and the National Volume Estimator Library (Wang 2015, NVEL 2016).

In 2015, breast height core samples were taken from all trees greater than 4 inches dbh in a subset of 3 plots per unit ($n = 9$ per treatment). Each tree was cored twice at breast height, the second core located approximately 90 degrees from the first. Increment borers were inserted beyond the sapwood-heartwood boundary, which was identified and labeled upon extraction. Associated bark widths were measured via bark gauges in a location on the bole in close proximity to each of the two cores extraction sites. Sapwood width and annual radial increment over the last five years were measured to the nearest 100^{th} of an inch using a binocular microscope and electronic digital calipers. Sapwood basal area (SA) and 5-year basal area increment were each calculated by referencing bark thickness and DBH. Basal area increment was calculated as both the average basal area increment ($\text{in}^2 \text{ year}^{-1}$) for the trees in each treatment

(BAI) as well as the total average basal area increment per acre (BAIac, $\text{ft}^2 \text{ acre}^{-1} \text{ year}^{-1}$). To avoid bias from smaller, suppressed trees in the control that were removed from the cut units, we also compared BAI of only the dominant and codominant trees in the CO units to the trees in the other treatments. Additionally, trees were divided into two age classes: tagged trees that were ≥ 6 in. dbh at the time of the initial harvest and ingrowth that has arisen since 1992 for comparison of the two cohorts.

Sapwood basal area was used to estimate total crown leaf area (LA) using species specific sapwood-leaf area prediction equations developed by Monserud and Marshall (1999) and O'Hara and Valappil (1995). Equations were provided for both ponderosa pine and Douglas-fir by Monserud and Marshall (1999). The Douglas-fir equations used crown ratio as a second coefficient, while ponderosa pine equations used crown length as the second coefficient. In the event that crown length was not available, equations by O'Hara and Valappil (1995) were used, as these required no additional coefficients aside from sapwood area. Equations by Monserud and Marshall (1999) were otherwise preferred because the sizes of sample trees used in their analysis were comparable to our sample trees. Leaf area for grand fir was calculated using a constant ratio developed by Waring et al. (1982). Leaf area of all trees were aggregated and expressed on a per unit area basis as leaf area index (LAI; $\text{m}^2 \text{m}^{-2}$) (Watson 1947). Average annual growth efficiency (GE) of trees was calculated using BAI ($\text{in}^2 \text{ year}^{-1}$) as the numerator and SA (in^2) as the denominator, as these were the two available metrics that were directly measured, minimizing extrapolation error (Seymour and Kenefic 2002, Woodall et al. 2003, McDowell et al. 2007).

2.2.4 Data analysis

When a mixed-effects linear model was fit to the response variables (TPA, BA, SDI, BAI, BAIac, LAI, and GE) the residuals for all of the response variables except BAI and BAIac exhibited non-normal distributions, inhibiting the use of linear modeling. Both exponential and log transformations were unable to normalize the residuals, with the exception of QMD₁₀₀. As a result, non-parametric permutation testing (Manly 2002) was used to analyze differences in response variables with several linear contrasts. First, an analysis of variance (ANOVA) was conducted with each response variable and treatment as the explanatory variable. Response variables were then randomly reassigned to each of the 12 units 999 times to create a sample of F-statistics that represent randomly allocated responses. To test the null model, i.e. the response variables are the same across treatments, the F-statistic from the original model was compared to the distribution of simulated F-statistics. If the F-statistic was unusually small relative to the simulated F-statistics, then the response variables were unlikely to have arisen if the null model is true, supporting an alternative hypothesis. This process was repeated using t-statistics from a simple linear model comparing each of the fuel reduction treatments to the control.

This application of permutation testing was used to examine several contrasts for each response variable. First, an analysis of variance was run to determine (1) whether the response variable in any of the treatments was different than any of the other treatments (CO-NB-SB/WB –FB/DB). Then simple linear modeling was applied to: (2) test whether treating a stand resulted in a different response irrespective of treatment type (CO-NB & SB/WB & FB/DB); (3) compare the response in each of the treatments to the control (CO-NB, CO- SB/WB, CO- FB/DB); (4) contrast responses in each of the burn treatments to the no burn treatment (NB- SB/WB, NB-FB/DB); (5) determine whether the response differed between the burned treatments versus the

no burn treatment (NB- SB/WB & FB/DB); and (6) check for differences in the response between the wet burns and dry burns (SB/WB - FB/DB).

2.3 Results

2.3.1 Stand structure

In the thinning, stem densities in the restoration treatment units were similar in 2015, with an average of 84 trees per acre (TPA) across treatments compared to a mean of 131 TPA in the control units (Table 2). Stem densities in the CO were significantly greater than any of the restoration treatments, with approximately 36% more TPA. Similarly, basal area (BA) was on average 27% greater in the CO relative to each of the restoration treatments, while stand density index (SDI) was about 28% greater ($p < 0.05$). Mortality was approximately 6% in the CO (28 TPA), and ranged from 11 to 16% (6-8 TPA) in the restoration treatments. Species composition was similar across treatments, with approximately 97% of the basal area composed of ponderosa pine. Douglas-fir comprised most of the remainder, with traces of lodgepole pine also present. QMD ranged from 13.1 inches in the CO to 14.8 inches in the FB (Table 2), and appeared to increase with treatment intensity. QMD₁₀₀ averaged at 17.5 inches across all treatments.

In the shelterwood, TPA was lower in each of the treated stands by 43-59% relative to the control, which had a density of 129 TPA (Table 2). BA and SDI were also lower in the restoration treatments relative to the control, by 39-47% and 43-59%, respectively ($p < 0.01$). Mortality was approximately 5% in the CO (29 TPA) and between 6-10% in the restoration treatments (8-12 TPA). Ponderosa pine constituted about 97% of the live BA in the WB and DB units, but in the NB and CO units, only accounted for 76-89% of the total species composition (Table 2). In the NB treatment, the other 23% were primarily small diameter Douglas-fir, which

were likely advanced regeneration that released after the initial harvest (Figure 2f). QMD was tended to increase with treatment intensity, ranging from 13.5 inches in the CO to 15.4 inches in the WB, while QMD₁₀₀ averaged 18.2 inches across treatments.

Table 2. Stand-level summaries for 2015, 23 years after treatment. Trees per acre (TPA), basal area per acre (BA), quadratic mean diameter (QMD) and stand density index (SDI) are listed for both the thinning and shelterwood by species (Ponderosa pine, PIPO; Douglas-fir, PSME). Differences in QMD were tested using only the tallest 100 trees per treatment (QMD₁₀₀); QMD for all trees > 4 inches dbh are listed here. Proportions of total for each species are included in parentheses, and alphabetical letters represent significant differences among treatments. All letters are uppercase, indicating significance at $p < 0.05$.

Thinning								Shelterwood							
	Trt	PIPO		PSME		Stand Total			Trt	PIPO		PSME		Stand Total	
TPA	CO	131	(0.97)	4	(0.03)	136	A	TPA	CO	94	(0.73)	35	(0.27)	129	A
	NB	86	(0.98)	2	(0.02)	88	B		NB	56	(0.77)	17	(0.23)	73	B
	SB	89	(0.95)	4	(0.05)	93	B		WB	50	(0.94)	3	(0.06)	53	B
	FB	78	(0.99)	1	(0.01)	79	B		DB	56	(0.92)	5	(0.08)	61	B
BA	CO	123	(0.96)	5	(0.04)	128	A	BA	CO	98	(0.76)	31	(0.24)	129	A
	NB	91	(0.98)	2	(0.02)	93	B		NB	70	(0.89)	9	(0.11)	79	B
	SB	89	(0.96)	3	(0.04)	93	B		WB	66	(0.97)	2	(0.03)	68	B
	FB	94	(0.99)	1	(0.01)	94	B		DB	69	(0.97)	2	(0.03)	71	B
QMD	CO	13.1		14.9		13.1		QMD	CO	13.8		15.0		13.5	
	NB	13.9		13.8		13.9			NB	9.3		10.8		14.1	
	SB	13.6		11.7		13.5			WB	15.1		10.0		15.4	
	FB	14.8		13.8		14.8			DB	15.6		12.7		14.6	
SDI	CO	196	(0.96)	8	(0.04)	203	A	SDI	CO	151	(0.75)	50	(0.25)	201	A
	NB	142	(0.98)	3	(0.02)	146	B		NB	105	(0.88)	15	(0.12)	119	B
	SB	141	(0.96)	5	(0.04)	146	B		WB	98	(0.97)	3	(0.03)	101	B
	FB	144	(0.99)	1	(0.01)	145	B		DB	104	(0.96)	4	(0.04)	108	B

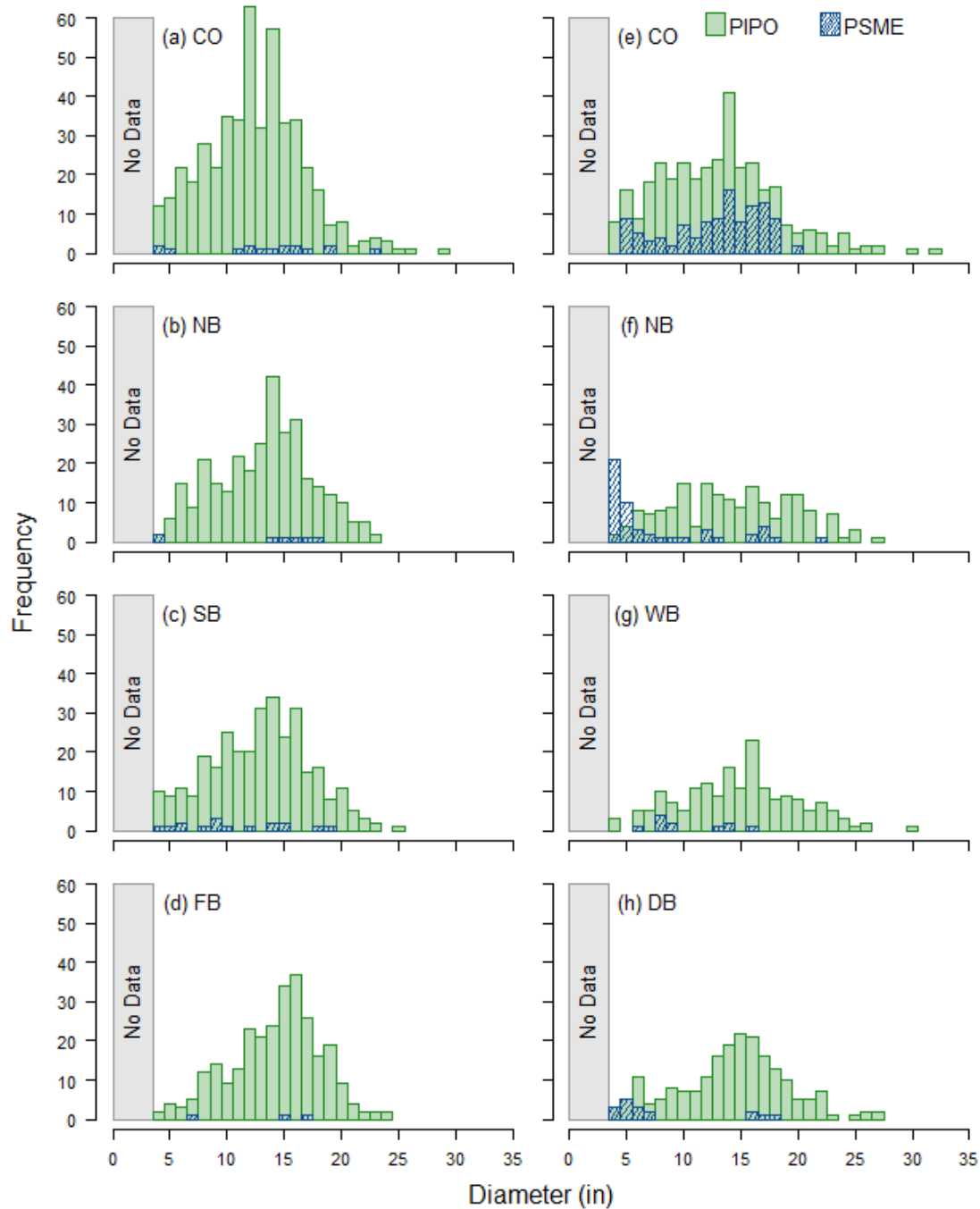


Figure 2. Diameter distributions by species (ponderosa pine, PIPO; Douglas-fir, PSME) for the Control, No Burn, Spring Burn, Fall Burn in the thinning (left column; a-d) and Control, No Burn, Wet Burn, Dry Burn in the shelterwood (right column; e-h) studies 23 years after harvest, overlaid by species. Only trees that are greater than 4 inches dbh are included in this distribution.

2.3.2 *Stand volumes*

In the thinning, pre-treatment volume was similar in all three of the restoration treatment units in 1991 ($p = 0.5$). Thinning resulted in similar volumes across restoration treatments, and in 1993, a year harvesting, volume in the restoration treatments was each significantly lower than the control ($p < 0.01$). This trend continued through 2005 ($p < 0.01$) and into 2015 ($p < 0.01$) (Figure 3a). In 2015, tree volume was $3,144 \text{ ft}^3 \text{ acre}^{-1}$ in the control, and between $2,200 \text{ ft}^3 \text{ acre}^{-1}$ (SB) and $2,357 \text{ ft}^3 \text{ acre}^{-1}$ (FB) in the restoration treatments. In the shelterwood, all three of the restoration treatments supported similar volumes in 1991 prior to harvesting ($p = 0.6$). Harvesting reduced volume in each of the three restoration treatments relative to the control ($p < 0.01$), but resulted in similar volumes across restoration treatments. This remained true through 2005 and into 2015 ($p < 0.007$ both years) (Figure 3c). Volume in 2015 was $4,102 \text{ ft}^3 \text{ acre}^{-1}$ in the control, and from $1,901 \text{ ft}^3 \text{ acre}^{-1}$ (DB) to $2,079 \text{ ft}^3 \text{ acre}^{-1}$ (NB) in the restoration treatments.

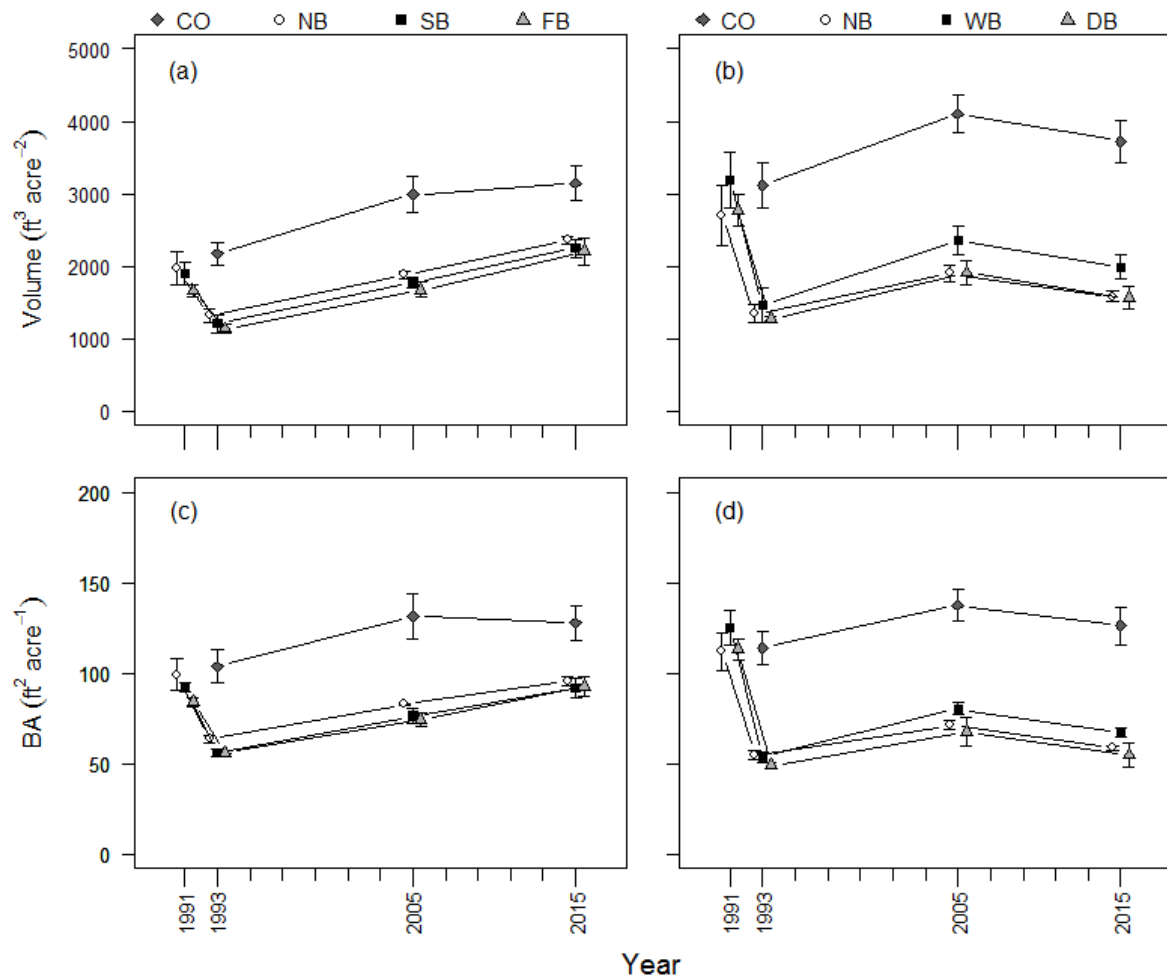


Figure 3. Volume (top row; a, b) and basal area (bottom row; c, d) trajectories for the thinning (left column; a, c) and shelterwood (right column; b, d) in each of the four treatments: Control (CO), No Burn (NB), Spring or Wet Burn (SB/WB) and Fall or Dry Burn (FB/DB). Harvesting was conducted in 1992 and treatments continued into 1994.

2.3.3 Basal area increment

In the thinning, individual tree basal area increment (BAI) ranged from 1.39 in² year⁻¹ (CO) to 2.23 in² year⁻¹ (DB). Treating the stand, regardless of treatment, resulted in greater BAI relative to the control ($p < 0.03$), but did not differ among restoration treatments (Table 3). When only the Dominant and Codominant trees in the CO were considered, BAI was still greater in the

NB and FB treatments ($p = 0.03$ and 0.01 , respectively) relative to the CO, but not in the SB treatment. Basal area increment per acre (BAIac) was similar across all treatments, ranging from 1.11 - $1.3 \text{ ft}^2 \text{ acre}^{-1} \text{ year}^{-1}$ (Table 3). In the shelterwood, only the WB treatment resulted in greater BAI relative to the control ($p = 0.02$), whether or not all trees in the CO were considered or only the Dominant/Codominant trees. BAI in the WB was also greater than in the NB treatment ($p = 0.06$). BAIac was also similar across treatments, ranging from 0.75 - $1.05 \text{ ft}^2 \text{ acre}^{-1} \text{ year}^{-1}$ (Table 3).

2.3.4 Leaf Area Index and Growth Efficiency

In the thinning, leaf area index (LAI) ranged from $1.8 \text{ m}^2 \text{ m}^{-2}$ in the NB to $2.44 \text{ m}^2 \text{ m}^{-2}$ in the CO but were not statistically different across treatments (Table 3). Growth efficiency (GE), calculated using BAI and sapwood area, was similar across treatments. In the shelterwood, burning treatments reduced LAI by 36% (DB) and 52% (WB) relative to the control ($p < 0.05$). The WB treatment also reduced LAI by 43% relative to the NB ($p = 0.03$). GE was the same across treatments, at $0.02 \text{ in}^2 \text{ in}^{-2}$ (Table 3).

Table 3. Average growth metrics from tree cores that were taken from a subsample of 36 plots in each installation. BAI ($\text{in}^2 \text{yr}^{-1}$) refers to the average annual growth of individual trees over the past 5 years. BAIac ($\text{ft}^2 \text{acre}^{-2} \text{year}^{-1}$) refers to average annual growth of all trees in the stand. LAI ($\text{m}^2 \text{m}^{-2}$) was calculated using published sapwood allometries (see text for details). GE was defined as in^2 of BAI per in^2 sapwood. Uppercase letter indicate significance at the 0.05-level; lowercase at the 0.1-level. Standard errors are in parentheses.

Thinning											
	CO			NB			SB			FB	
BAI	1.39	(0.05)	A	2.10	(0.10)	B	2.02	(0.10)	B	2.23	(0.10) B
GE	0.02	(0.001)	a	0.02	(0.002)	ab	0.02	(0.001)	b	0.02	(0.002) ab
BAIac	1.30	(0.10)		1.22	(0.17)		1.19	(0.13)		1.11	(0.03)
LAI	2.44	(0.35)		1.79	(0.28)		1.99	(0.26)		1.82	(0.20)
Shelterwood											
	CO			NB			WB			DB	
BAI	1.53	(0.07)	A	2.20	(0.15)	a	2.54	(0.18)	Bb	2.48	(0.13) ab
GE	0.02	(0.005)		0.02	(0.002)		0.02	(0.004)		0.02	(0.002)
BAac	0.96	(0.12)		1.05	(0.18)		0.75	(0.12)		0.89	(0.06)
LAI	2.27	(0.18)	A	1.91	(0.27)	AB	1.08	(0.30)	C	1.44	(0.08) bC

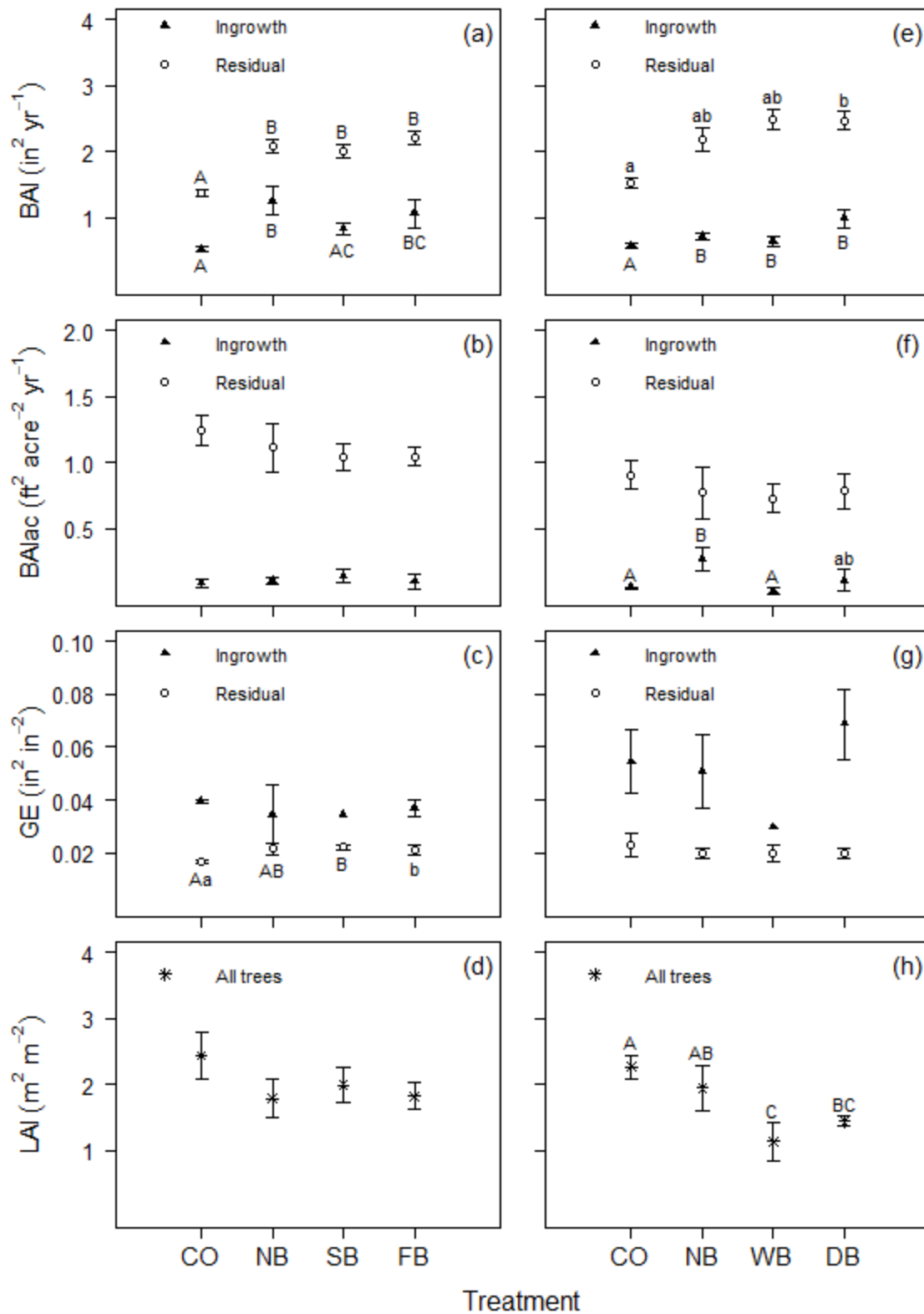


Figure 4. Average basal area increment per tree (BAI), basal area increment per acre (BAIac), and growth efficiency (GE) in 2015 in the thinning (left column; a-c) and shelterwood (right column; e-g) are partitioned into residual overstory trees from the initial harvest and ingrowth

that has developed since then. Leaf area index (LAI) is considered for all trees > 4 inches dbh (bottom row; d,h)

2.3.5 Regeneration

In the thinning, seedling and sapling density was lower in the CO units relative to the other treatments by 62-71%, and greatest in the WB, with an average of 774 TPA (Table 4). The discrepancy between restoration treatments and the control seemed primarily due to the absence of a large cohort of recent regeneration in the control—less than 2 ft. tall—that was present in each of the other treatment (Figure 5a-d). Seedling and sapling composition was about 64% ponderosa pine and 36% Douglas-fir across treatments (Table 4).

In the shelterwood, seedling and sapling density was highest in the CO, with 2944 stems per acre. Restoration treatments tended to have lower seedling/sapling densities, ranging from 1305 (DB) to 2184 (NB) stems per acre. Douglas-fir accounted for up to 90% of the density in the CO and 76% of density in the NB. In contrast, Douglas-fir only accounted for about half of seedling/saplings in the two burned treatments (Table 4). Density was highest in the CO, followed by the WB, NB, and DB, respectively. In contrast to the thinning study, the greatest frequency of regeneration in the smallest height classes (< 2 ft.) was in the control (Figure 5e). Also notable were the predominance of Douglas-fir trees 5 to 15 ft. tall in the NB treatment (Figure 5f).

Table 4. Seedling/sapling densities (TPA) for each of the treatments within both the thinning and shelterwood studies. Percentages of total are in parentheses. In the shelterwood, grand fir, lodgepole pine, and subalpine fir accounted for 0-60 seedlings per acre. Alphabetical letters represent significant differences in total seedling/sapling density by treatments, with uppercase letters indicating significance at the 0.05-level and lowercase at the 0.1-level.

Thinning							
	Trt	PP		DF		Stand Total	
TPA	CO	161	(0.73)	61	(0.27)	222	A
	NB	380	(0.65)	201	(0.35)	581	ab
	SB	431	(0.56)	343	(0.44)	774	B
	FB	378	(0.61)	239	(0.39)	617	ab
Shelterwood							
	Trt	PP		DF		Other	Stand Total
TPA	CO	222	(0.08)	2662	(0.90)	60 (0.02)	2944
	NB	459	(0.24)	1479	(0.76)	3 (0.00)	1941
	WB	1207	(0.56)	947	(0.44)	0 (0.00)	2154
	DB	527	(0.40)	777	(0.60)	1 (0.00)	1305

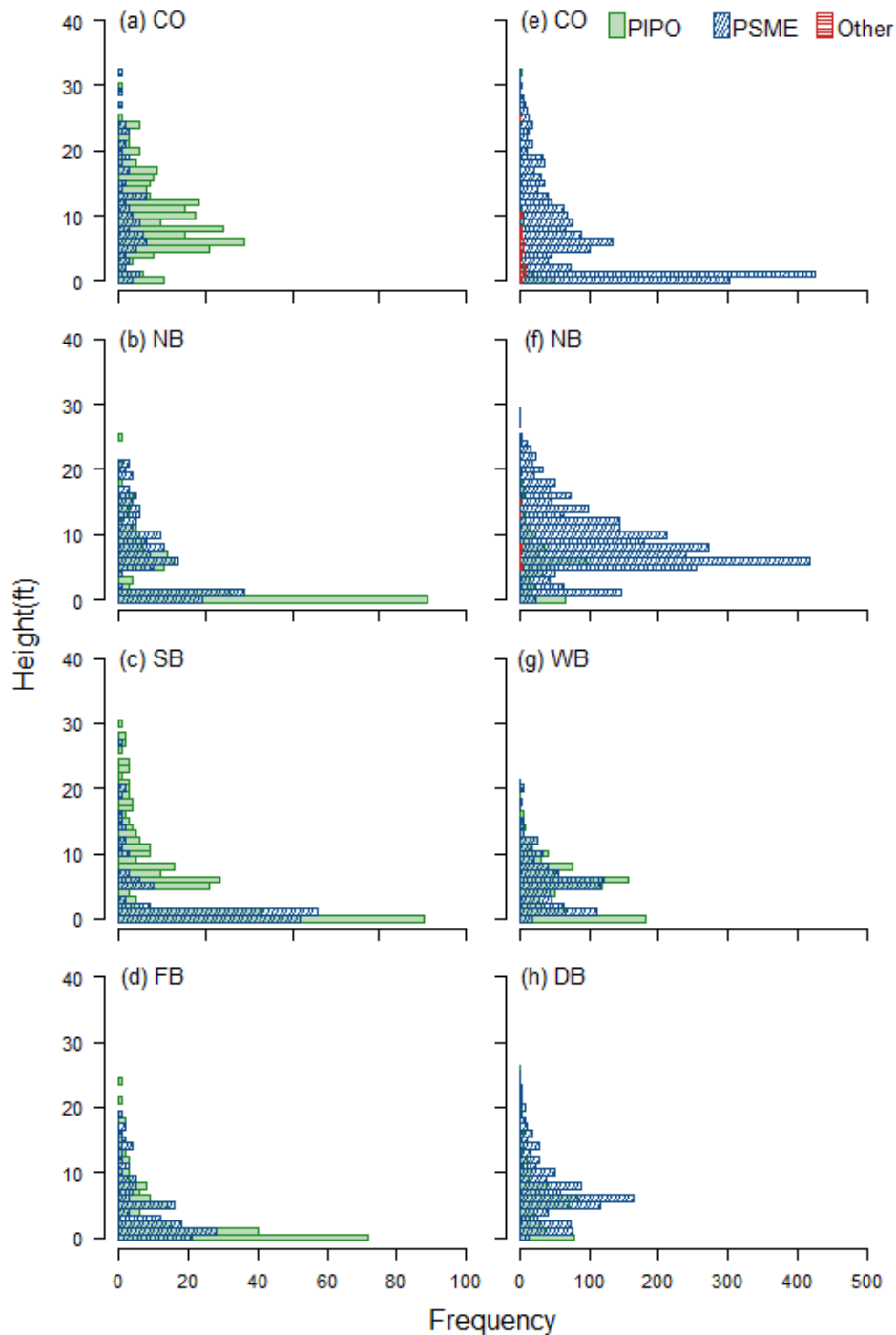


Figure 5. Height distributions for seedlings/saplings in the control (CO), no burn (NB), spring or wet burn (SB/WB) and fall or dry burn (FB/DB) treatments by species (Ponderosa pine, PIPO; Douglas-fir, PSME) in the thinning (left column; a-d) and shelterwood (right column; e-h), overlaid by species. Note the difference in frequency scales between studies; the shelterwood had a much higher density of seedlings and saplings than the thinning. “Other” represents < 2% of *Abies grandis*, *Pinus contorta*, and *Abies lasiocarpa*.

2.4 Discussion

2.4.1 Stand structure and composition

Twenty-three years post-treatment, overstory composition and densities were still similar to initial post-harvest conditions, with no differences among the three fuel reduction treatments. Similarity in density responses between cut-only and cut-and-burn treatments agrees with previous findings in the thinning (Sala et al. 2005), as well as similar studies in northeastern Oregon (e.g. Youngblood et al. 2006) and California mixed-conifer (e.g. Stephens and Moghaddas 2005), indicating fire-induced mortality has not been substantial in any of these studies. Mortality rates were similar across fuel reduction treatments in the thinning through 2015, though slightly higher than the first four years after treatment (Smith et al. 1999). Other studies have observed increased mortality from fall burns (e.g. Harrington 1987, Thies et al. 2005), attributed to increased severity of fall burns, or from spring burns, attributed to susceptibility to heat damage from low carbohydrate reserves during the growing season (Hough 1968, Garrison 1972, Swezy and Agee 1991, Harrington 1993). Greater mortality rates are also often attributed to greater fire intensity (e.g. Thies et al. 2005), and thus, in the shelterwood, the DB might have been expected to exhibit higher mortality rates compared to the WB. During the first 5 years after treatment, mortality from fire and other origins was 14% in the DB (for trees > 7 in. dbh) and 10% in the WB (Smith et al. 1999). In 2015, mortality was 6% of the total density in the DB and 9% in the WB, and at least partially attributed to the presence of comandra blister rust (*Cronartium comandrae* Pk.). Fire-induced mortality typically occurs only in the first few years after a fire, so it not surprising that we observed little differences in mortality 23 year post-fire. Similarities between initial post-fire mortality in the DB and WB was likely due to similar duff consumptions across burns, despite fuel moisture content differences at the time of burning.

Duff consumption averaged 1.1 inches in the WB and 1.4 inches in the DB, indicating that the WB and DB treatments may not have achieved the desired discrepancy in burning prescriptions (Smith et al. 1999).

2.4.2 Growth trends

Estimates of incremental growth (BAI) indicate that trees in the fuel treatments are still growing faster than trees in the control in both the thinning and shelterwood, even when only the dominant and codominant trees in the CO were considered. In the thinning, average basal area increment per tree (BAI) tended to be greater in the restoration treatments relative to the control, but did not differ among restoration treatments, similar to other findings in the Northern Rockies (Fiedler et al. 2010). In the shelterwood, average BAI also tended to be greater in all three fuel treatments relative to the control, although only the difference between the WB and CO was statistically significant. Increased BAI has been linked to greater individual tree resistance against bark beetles and stress (Kolb et al. 2007, McDowell et al. 2007, Hood et al. 2016). Similarly, larger trees have demonstrated greater resistance to fire-induced mortality, as larger diameter trees are able to withstand relatively greater cambial heating and tend to be associated with greater bark thickness (Martin 1965, Ryan 1982). Resistance of individual trees to disturbance agents is an important element in ecosystem resilience, which is often a key objective in many restoration efforts. Our results suggest that overstory trees in all of the treated stands are more resistant to disturbance agents such as bark beetles and wildfire relative to trees in the control, even after more than 20 years since treatment.

Additionally, annual stand-level growth (BAI_{ac}) was the same across treatments, indicating that the fewer trees in the treated units are accumulating just as much basal area growth per acre as the many trees in the control. This is similar to a thinning study of old ponderosa pine stands

in eastern Oregon, which tracked BAI every year for 6-14 years after treatment and found increases in average tree BAI, but not stand-level basal area increment (Latham and Tappeiner 2002). Increasing individual tree growth while maintaining stand-level volume accumulation allows for the ability of managers to achieve both restoration and timber production objectives.

In the thinning, individual tree BAI was similar whether or not burning was applied after thinning. This supports findings from a study in northern Arizona, where differences in tree growth between treated stands and the control were largely explained by reduced competition from thinning, and prescribed fire did not greatly contribute to increases in growth (Zausen et al. 2005). In a different study in Southwestern ponderosa pine, prescribed burning resulted in an initial decrease in growth rates for 2 years after treatment relative to an unburned stand, but after 2 years growth rates were similar across treatments (Sutherland et al. 1991). In our study, there was also no difference in growth between trees in the spring and fall burns, contrary to predictions that dormant season (fall) burns may limit a tree's ability for nutrient uptake (Hamman et al. 2008). Additionally, precipitation and nutrient leaching after late season burns may also limit nutrient availability after a fall burn (Huffman et al. 2001). However, it is likely that these effects – if they had occurred – would have diminished after 20 years, and our basal area increment measurements were based on only the most recent 5 years, thus not capturing potential immediate post-treatment effects. Additionally, the greatest impacts of burn seasonality may primarily pertain to understory composition, as these species tend to be sensitive to burning during different annual growth periods (Platt et al. 1988, Kauffman and Martin 1990, Kerns et al. 2006).

2.4.3 Regeneration

Differences in site productivity seemed to be a primary driver of differing regeneration responses between the thinning and shelterwood. Although we did not quantify site index in the two locations, empirical evidence suggests that the shelterwood is located in a more productive site. The shelterwood is situated at the base of the drainage, and is in the vicinity of mesic species, including grand fir, Engelmann spruce (*Picea engelmannii* Parry ex. Engelmann), and subalpine fir. Conversely, the thinning is upslope and lacks presence of mesic parent trees, reducing recruitment of Douglas-fir. Furthermore, logging history is also likely to have a great impact on species composition, stand structure, and fire behavior (Naficy et al. 2010). The stand in which the thinning is located has been repeatedly cut over the past century, potentially removing much of the Douglas-fir overstory and reducing subsequent Douglas-fir regeneration. Conversely, the shelterwood has not been harvested since the initial clearcut in 1906, and had a much greater proportion of Douglas-fir regeneration, especially in the CO and NB treatments (Gruell et al. 1982).

In the thinning, seeding and sapling density was greatest in the SB, followed by the FB and NB treatments. Pine density was similar across all three restoration treatments, while proportionally it was greatest in the CO. Given the absence of the smaller cohorts in the CO however, it is probable that the CO has actually experienced relatively little recruitment, and many of the trees < 4 in. dbh are older, suppressed trees. This highlights the uncertainty associated with the lack of age data, as we were unable to distinguish between advanced regeneration and new recruitment. For example, total regeneration densities in the shelterwood CO appeared to be almost 150% of those in the treatments, however, this is quite possibly due to

a greater density of older, suppressed trees in these units. Without age data it is impossible to determine the density of recruitment since time of treatment.

In the shelterwood, seedling and sapling densities were greatest in the CO, intermediate in the NB and WB, and lowest in the DB. Pine recruitment was greatest in the WB treatment, with more than double the density of any other treatment. Proportion of pine recruitment was only slightly lower in the DB relative to the WB, however, overall seedling/sapling density in the DB was almost half that of the WB. Negligible pine recruitment in both the NB and CO treatments indicates the necessity of broadcast burning to remove unwanted understory trees and prepare the seedbed for pine regeneration. High densities of Douglas-fir seedlings and saplings in the NB and CO treatments indicate that these treatments were not effective at perpetuating pine-dominated forests, unlike the two burn treatments. In all treatments, various levels of intervention will be required to maintain the reduced stand densities and open canopy conditions required for continued pine establishment and development.

2.5 Conclusion

The Lick Creek studies provide a unique opportunity to compare long-term treatment responses across two sites with different treatment histories, site productivities, and topographic locations while holding climate and geographic variables constant. They contribute to a growing body of research on the effects of restoration treatments on stand dynamics in the Northern Rocky Mountains. The Lick Creek installations are also distinctive in their longevity, as few studies are able to quantify treatment responses more than 20 years after harvest, especially in low-elevation, dry forest types. Our results indicate that even after 23 years, treatments are effective at maintaining reduced stand densities. In the thinning installation, where productivity

is low, all fuel reduction treatments resulted in similar stand structure, growth rates, and regeneration patterns. Treating this forest with any of the three treatment options would be preferable to a no-management alternative. In the shelterwood, where productivity was high, treatments that included prescribed fire demonstrated the greatest rates of growth, greatest proportion of pine regeneration, and perpetuation of fire-resilient stand characteristics.

Appendices

Appendix A. F and t statistics are reported for the ANOVA or linear model used to test contrasts. P-values from the nonparametric permutation testing are in parentheses. *** indicates significance at the 0.01 level; ** indicates significance at the 0.05 level; * indicates significance at the 0.1 level.

Thinning Stand Metrics											
Contrasts	TPA			BA			SDI			QMD	
CO - NB - SB - FB	3.72	(0.061)	*	7.75	(0.021)	**	6.72	(0.021)	**	1.00	(0.420)
CO & NB & SB & FB	10.52	(0.017)	**	23.20	(0.005)	***	20.15	(0.005)	***	0.98	(0.363)
CO - NB	-2.57	(0.029)	**	-3.97	(0.008)	***	-3.67	(0.008)	***	0.72	(0.513)
CO - SB	-2.30	(0.047)	**	-4.01	(0.007)	***	-3.63	(0.008)	***	0.15	(0.867)
CO - FB	-3.08	(0.017)	**	-3.81	(0.007)	***	-3.69	(0.007)	***	1.56	(0.158)
NB - SB & FB	0.02	(0.905)		0.01	(0.955)		0.00	(0.994)		0.02	(0.873)
NB - SB	0.07	(0.783)		0.00	(0.969)		0.00	(0.966)		0.33	(0.556)
NB - FB	0.56	(0.460)		0.05	(0.846)		0.00	(0.971)		0.33	(0.216)
SB - FB	0.62	(0.461)		0.04	(0.864)		0.00	(0.968)		2.00	(0.181)

Shelterwood Stand Metrics											
Contrasts	TPA			BA			SDI			QMD	
CO - NB - WB - DB	11.76	(0.003)	***	22.98	(0.004)	***	22.20	(0.006)	***	0.11	(0.942)
CO & NB & WB & DB	33.18	(0.001)	***	66.81	(0.002)	***	64.59	(0.003)	***	0.04	(0.816)
CO - NB	-3.94	(0.011)	**	-5.86	(0.010)	***	-5.79	(0.011)	**	0.38	(0.732)
CO - WB	-5.38	0.000	***	-7.28	(0.001)	***	-7.19	0.0000	***	0.24	(0.828)
CO - DB	-4.79	(0.001)	***	-6.88	(0.002)	***	-6.71	(0.003)	***	-0.14	(0.894)
NB - WB & DB	1.75	(0.211)		1.98	(0.201)		1.79	(0.220)		0.15	(0.701)
NB - WB	2.06	(0.177)		2.00	(0.196)		1.96	(0.196)		0.02	(0.883)
NB - DB	0.02	(0.912)		0.13	(0.747)		0.06	(0.828)		0.28	(0.636)
WB - DB	0.34	(0.600)		0.15	(0.724)		0.23	(0.650)		0.15	(0.754)

Appendix B. F and t statistics are reported for the ANOVA or linear model used to test contrasts for each growth metric. P-values from the nonparametric permutation testing are in parentheses. *** indicates significance at the 0.01 level; ** indicates significance at the 0.05 level; * indicates significance at the 0.1 level.

Thinning Growth Metrics									
Contrasts	BAI			BALac			LAI		GE
CO - NB - SB - FB	7.06	(0.026)	**	0.41	(0.731)		1.17	(0.379)	1.45 (0.321)
CO & NB & SB & FB	18.21	(0.004)	***	0.82	(0.398)		3.20	(0.121)	3.33 (0.116)
CO - NB	3.59	(0.006)	***	-0.45	(0.665)		-1.66	(0.161)	1.28 (0.261)
CO - SB	2.57	(0.037)	**	-0.67	(0.469)		-1.14	(0.261)	2.06 (0.076) *
CO - FB	4.28	(0.003)	***	-1.09	(0.314)		-1.58	(0.152)	1.12 (0.298)
NB - SB & FB	0.04	(0.840)		0.25	(0.626)		0.12	(0.739)	0.13 (0.697)
NB - SB	1.04	(0.319)		0.8	(0.842)		0.27	(0.587)	0.62 (0.448)
NB - FB	1.92	(0.209)		0.6	(0.554)		0.04	(0.853)	0.40 (0.555)
SB - FB	2.92	(0.136)		0.17	(0.678)		0.19	(0.663)	0.89 (0.383)
Shelterwood Growth Metrics									
Contrasts	BAI			BALac			LAI		GE
CO - NB - WB - DB	3.09	(0.080)	*	1.15	(0.377)		5.52	(0.031)	** 0.83 (0.477)
CO & NB & WB & DB	5.10	(0.043)	**	0.32	(0.569)		9.47	(0.020)	** 0.05 (0.818)
CO - NB	0.87	(0.424)		0.47	(0.659)		-1.13	(0.312)	0.89 (0.386)
CO - WB	2.90	(0.021)	**	-1.27	(0.249)		-3.78	(0.003)	*** -0.67 (0.535)
CO - DB	1.76	(0.120)		-0.40	(0.692)		-2.63	(0.044)	** 0.31 (0.803)
NB - WB & DB	2.87	(0.131)		2.43	(0.172)		5.77	(0.043)	** 1.51 (0.226)
NB - WB	4.15	(0.062)	*	3.14	(0.121)		7.06	(0.029)	** 2.41 (0.129)
NB - DB	0.02	(0.893)		0.00	(0.956)		0.04	(0.847)	0.05 (0.852)
WB - DB	1.30	(0.260)		0.71	(0.410)		1.33	(0.260)	0.94 (0.380)

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SECTION 3:

Long-term effects of fuel treatments on aboveground biomass accumulation in ponderosa
pine forests of the Northern Rocky Mountains

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Abstract

In western North America, many low-elevation, dry forest types historically experienced frequent, low-severity fires. However, European settlement and fire suppression policies have contributed to over a century of fire exclusion, substantially altering forest structure and composition. Specifically, many forests have experienced considerable increases in forest biomass relative to pre-settlement conditions. As a result, carbon storage in these forests have partially offset rises in anthropogenic carbon emissions over the past century. However, it is becoming more difficult to exclude fire from the landscape, as changes in climate are resulting in longer and drier summers, and an excess of forest fuels are present across much of the landscape. High-severity fires are growing in frequency, magnitude, and duration, resulting in substantial carbon offset reversals. Fuel treatments aim to restore low-severity fire conditions by strategically reducing available forest fuels. However, it is unclear as to how different fuel

treatments impact aboveground biomass, especially in the long-term. This research evaluated aboveground biomass responses 23 years after treatment in two silvicultural installations with different underburning prescriptions. A thinning and shelterwood were harvested in 1991 at different locations in the same drainage in southwestern Montana. Following harvesting, three burning prescriptions were applied in each installation. In the thinning, burning prescriptions included a fall burn, a spring burn, and a no-burn treatment. In the shelterwood, burning included a wet burn, dry burn, and no-burn treatment. Across all fuel treatments, tree biomass had recovered to pre-harvest levels by 2015. In the thinning, total aboveground and live-tree biomass were greatest in the control and did not differ among fuel treatments. Forest floor biomass was lower in the two burned treatments relative to the two unburned treatments. Seedling, vegetation, stump, and snag biomass did not differ among any of the four treatments. In the shelterwood, total aboveground and live-tree biomass were both greater in each of the unburned treatments relative to the burned treatments. Forest floor biomass also tended to be lower in the burned treatments, along with snag biomass. Seedling, vegetation, and stump biomass were similar across all treatments. This research indicates that tree biomass can recover to pre-harvest levels in under 23 years, while still maintaining reduced stand densities that promote restoration objectives. However, proliferations of seedling biomass indicate that understory treatments are currently needed to maintain treatment effectiveness at reducing fire severity.

3.1 Introduction

In the western United States, many low-elevation, dry forest types have experienced substantial increases in forest biomass over the past century. For instance, between 1953 and 2012, net forest volume in the Intermountain West increased by 30 percent (Oswalt et al. 2014). This increase in biomass is primarily attributed to declines in logging activity and the rise of fire

exclusion policies. Prior to European settlement, western dry forests experienced low to mixed severity fires on average intervals of 3 to 50 years (Hessburg and Agee 2003, Fitzgerald 2005). However, logging, grazing, and fire suppression have all contributed to interruptions in natural disturbance regimes (Pyne 1982, Agee 1993, Stephens and Ruth 2005, van Wagtendonk 2007, Naficy et al. 2010). It is estimated that of the 236 million acres in the West identified as forestlands, 67 million acres have been moderately or significantly altered by fire exclusion (Rummer et al. 2005).

One inadvertent benefit of the suspension of natural disturbance regimes and increases in forest biomass has been a net gain in carbon sequestration throughout many of the dry forest types in the West, which has helped to offset fossil fuel emissions (Sohnngen and Haynes 1997, Houghten et al. 2000, Hurtt et al. 2002). However, as changing climate conditions result in longer and drier fire seasons, fire exclusion is becoming less feasible, and 100 years of fuel accumulation is resulting in much higher severity, intensity, and magnitude of fires than were historically present on the landscape (Westerling et al. 2006, Flannigan et al. 2013, van Mantgem et al. 2013). Carbon emitted from high-severity forest fires can be substantial, and in some years, even exceed regional annual carbon emissions from fossil fuels (Wiedinmyer and Neff 2007, Dore et al. 2008). This transformation of a forest from a sink to a source is known as an offset reversal.

Wildfire hazard is typically mitigated by applying a diverse set of silvicultural, mechanical, and prescribed fire strategies to alter the quantity and structure of forest fuel complexes, and thereby alter potential fire behaviors (Graham et al. 2004). For instance, crown fire hazard is usually addressed by reducing canopy density, removing small diameter ladder fuels, and increasing canopy base height (Graham et al. 1999, Keyes and O'Hara 2002, Agee and

Skinner 2005, Fitzgerald 2005, Reinhardt et al. 2008). Activity fuels produced by harvesting are often treated either by piling and burning, mechanical application, or when allowable, broadcast burning (Agee and Skinner 2005). It remains unclear as to how these fuel treatments impact ecosystem biomass and whether or not the net amount of carbon released from fuel reduction treatments is less than that of potential wildfire emissions. However, there is increasing evidence that reducing wildfire severity, even at the cost of initial carbon reductions from thinning, will result in a more sustainable carbon sink over the long term (Hurteau et al. 2008, Hurteau and North 2009, Amiro et al. 2010, Dore et al. 2010). For example, Hurteau et al. (2008) found that thinning prior to wildfire occurrence could reduce carbon emissions from live tree biomass by as much as 98%. As wildfire is an inevitability across much of the western landscape, implications of these treatments for offsetting carbon emissions may be substantial.

In addition to reducing emissions from catastrophic wildfires, fuel treatments may help restore attributes of pre-settlement forest structure. There is evidence that the fewer large trees present in western forests prior to European settlement stored more carbon than the abundance of smaller trees that currently exist (Fellows and Goulden 2008). By reducing competition for water, nutrients, and light, residual trees may experience increased photosynthetic rates after harvest, increasing the amount of carbon sequestered per individual tree (Feeney et al. 1998, Skov et al. 2004, Sala et al. 2005, Dore et al. 2010). Increased vigor of individual trees may also contribute to maintaining long-term live biomass carbon sinks in the face of increasing occurrences of drought and other disturbances such as insects and disease (Larsson et al. 1983, Skov et al. 2004, Hood et al. in press).

It is estimated that of the 6.9 billion bone dry tons of standing timber volume in the 15 western states of the United States, removal of approximately 2 billion bone dry tons (almost

30%) is required from forests with historically mixed- and low- severity fire regimes in order to restore pre-settlement biomass quantities (Rummer et al. 2005). However, there is uncertainty regarding where and how this biomass should be removed to maximize carbon storage while also minimizing emissions from wildfires and meeting additional ecological restoration objectives. Moreover, it is unclear how biomass will respond to these treatments in the long-term, as very few studies have examined biomass over durations longer than 3 years post-treatment.

This study evaluated aboveground biomass in a ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) /Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.)) forest in western Montana 23 years after applications of several common fuel reduction treatments using different cutting and prescribed burning prescriptions. By quantifying aboveground biomass responses to treatments, our goal was determine how treatments impact each of the different components of aboveground biomass, and to describe the development of these components over a 23-year time period. This research informs carbon models on the long-term effects of fuel reduction treatments on the structure of aboveground biomass in Northern Rocky Mountain ponderosa pine forests. We hypothesized that 23 years after treatment (1) tree biomass would remain lower in all fuel reduction treatments relative to pre-treatment biomass levels, as well as the untreated control, while (2) forest floor biomass and understory vegetation biomass would have recovered since time of treatment and be consistent across all treatments. We also predicted that (3) regeneration biomass would be greatest when thinning without underburning is applied, and that (4) snag biomass would be greatest in the untreated control. Finally, we expected (5) total aboveground biomass in each of the fuel reduction treatments would continue to be lower than the control 23 years after treatment.

3.2 Methods

3.2.1 Study site

Research was conducted at the Lick Creek Demonstration/Research Forest (hereafter: Lick Creek) on the Darby Ranger District of the Bitterroot National Forest in southwestern Montana (46°5'N, 114°15'W) (Figure 1a). The site is semi-arid, with an estimated average annual temperature of 7 °C and precipitation of 400 mm, with about 30% of this annual precipitation falling as snow (Gruell et al. 1982, DeLuca and Zouhar 2000). Elevations within Lick Creek range from approximately 1300 to 1500 meters, with slopes primarily ranging from 0 to 30 percent (Menakis 1994). Soils are relatively shallow or moderately deep, and are classified as Elkner Gravelly Loam, coarse-loamy, mixed, frigid Typic Cryochrepts, with highly weathered granite parent material (DeLuca and Zouhar 2000).

Overstory vegetation consists principally of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson var. *ponderosa* C. Lawson), with Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.)) in the understory, although grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*), and lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *latifolia* Engelm. ex S. Watson) are also intermittently present. Habitat types as classified by Pfister et al. (1997) within the drainage are Douglas-fir/snowberry (*Symphoricarpos albus* (L.) S.F. Blake) and Douglas-fir/pinegrass (*Calamagrostis rubescens* Buckley) located on the southerly aspects, and Douglas-fir/dwarf huckleberry (*Vaccinium caespitosum* Michx.), blue huckleberry (*Vaccinium globulare* Douglas ex Torr.), twinflower (*Linnaea borealis* L. subsp. *americana* (Forbes) Hultén ex R.T. Clausen) and grand fir/twinflower on the northwest aspects (Menakis 1994).

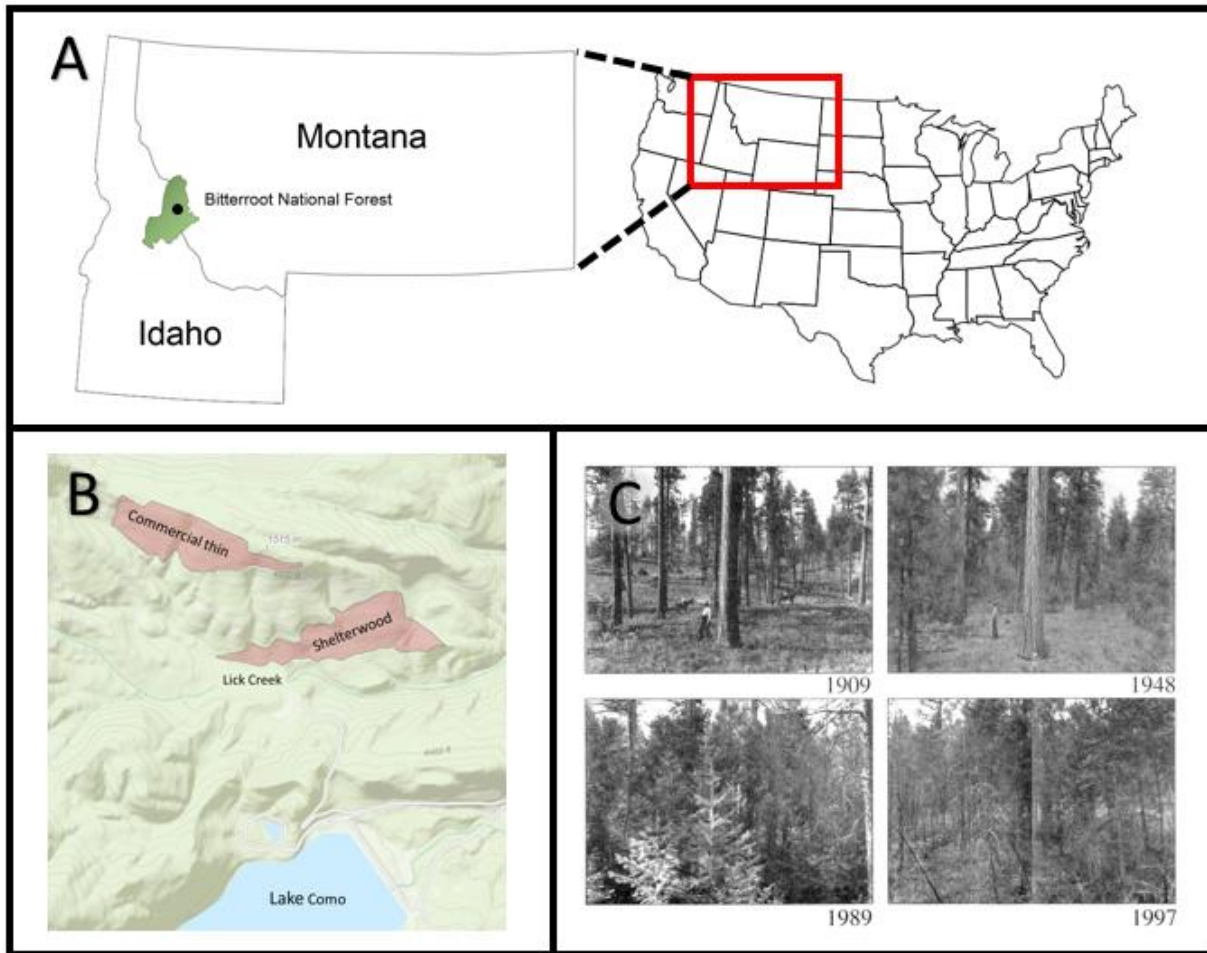


Figure 1. (a) The study site (black circle) is located on the Bitterroot National Forest in western Montana near the Idaho-Montana state border. (b) The two silvicultural installations are located between 4300 and 5000 ft. in elevation on south facing slopes in the Lick Creek drainage. The shelterwood is located downslope near Lick Creek, while the thinning is upslope, in proximity to the ridge. (c) The Lick Creek drainage is the location of the well-known photo series documenting forest succession after the first large USFS timber sale in ponderosa pine in 1907. Photos have been taken approximately every decade since 1909 at established photo points (Gruell et al. 1982).

3.2.2 Experimental design

There are two silviculture installations examined: a commercial thinning targeting a residual basal area of $12 \text{ m}^2 \text{ ha}^{-1}$ and a retention shelterwood aiming to reduce basal area to $9 \text{ m}^2 \text{ ha}^{-1}$ (Table 1). Both prescriptions were harvested in July and August of 1992. The thinning is located upslope of the Lick Creek drainage, with a southerly aspect and elevations of 1460 to

1540 meters (Figure 1b). The thinning had a pre-treatment stand age of 70 years, with approximately 369 trees ha⁻¹, 19-23 m² of basal area (BA) per hectare, and a 93% ponderosa pine species composition. The 1992 thinning resulted in an average of 219 trees ha⁻¹ and BA of 14 m² ha⁻¹. In order to examine the effects of burning and burn seasonality, three units were burned in the fall of 1993 (FB), three units were burned in the spring of 1994 (SB), and three units were left unburned (NB). Three additional units serve as an untreated control (CO) (Smith et al. 1999).

The shelterwood is positioned towards the base of the drainage, with a primarily southerly aspect and elevations of 1320 to 1390 meters (Figure 1b). Prior to the 1992 cutting, the 85 year old stand supported 435 trees ha⁻¹, 27 m² ha⁻¹ BA, and a 72% ponderosa pine species composition. The shelterwood cutting resulted in a post-harvest density of 174 trees ha⁻¹ and 12 m² ha⁻¹ basal area. In addition to the overstory cutting, thinning was also applied to several dense pockets of smaller trees in three of the fuel reduction treatment units in order to reduce sapling density. Post-harvest burning treatments in the shelterwood consisted of a high consumption burn (lower duff was dry; DB) in three of the 12 units, a low consumption burn (lower duff was wet; WB) in three units, and a no burn treatment (NB) in three cut units. Three units were served as an untreated control (CO).

Harvesting was conducted in 1992 in both the shelterwood and thinning units using chainsaw felling, followed by winch yarding to selected trails and skidding by a crawler tractor. Aside from the tree tops, which were cut at 15 cm diameter (outside bark), all other limbs were yarded to the landing with the bole and removed at the roadside landing (Smith et al. 1999). Burning occurred in the spring and fall of 1993, as well as the spring of 1994 (Table 1).

Table 1. Summary of the two silvicultural studies: a commercial thinning and a retention shelterwood, with each of the different levels of underburning applied to each study implemented between 1992 and 1994 (Smith et al. 1999). Pre-treatment values in the Control are unavailable as they were not measured prior to treatment.

Thinning							
Treatment	Abbreviation	Pre-treatment		Cutting target	Fuels treatment	Post-treatment	
		TPH	BA	(m ha ⁻¹ BA)		TPH	BA
Control	CO	-	-	None	None	454	24
Thin and No Burn	NB	384	21	12	None	220	13
Thin and Wet Burn	SB	435	20	12	1994 Spring burn	310	13
Thin and Dry Burn	FB	447	23	12	1993 Fall burn	279	15

Shelterwood							
Treatment	Abbreviation	Pre-treatment		Cutting target	Fuels treatment	Post-treatment	
		TPH	BA	(m ha ⁻¹ BA)		TPH	BA
Control	CO	-	-	None	None	728	26
Cut and No Burn	NB	534	29	9	None	244	11
Cut and Wet Burn	WB	672	26	9	May 1993 low consumption burn	179	12
Cut and Dry Burn	DB	677	26	9	May 1993 high consumption burn	238	13

3.2.3 Sampling design

All trees and saplings were measured on a systematic grid of 12, 0.04-ha permanent circular plots located within each of the 12 treatment units in 1993 (post-harvest), 2005, and 2015. Pre-harvest (1991) data was also collected in the 9 fuel reduction treatment units, but not in the control units. Species, diameter (dbh), total height, crown base height, crown ratios, crown position, and health status were recorded for all trees greater or equal to 10 cm dbh. Species, diameter, crown ratio, and a subset of heights were measured on all saplings, which were defined as all trees greater or equal to breast height (1.4 m), but less than 10 cm dbh. In 2015, seedlings,

forest floor biomass, understory vegetation, and stumps. Seedlings, which included all regeneration of less than 1.4 m tall, were measured on a 0.004-ha subplot, nested within the 0.04-ha plot. Species and height class were recorded for all seedlings, where heights were categorized by bins centered at 0.06 m, 0.3 m, 0.6 m, 0.9 m, and 1.2 m. Forest floor biomass—litter and duff, fine woody biomass, and coarse woody biomass—were measured along two, 16.7 m transects per plot using line intercept sampling methods developed by Brown (1974). The first transect rotated sequentially from upslope to downslope by 45° (i.e. upslope for plots 1,5,9; 45° from upslope for 2,6,10; 90° from upslope for 3,7,11; and 135° for plots 4,8,12). The second transect was located 90° from the first transect. Live and dead shrub and herb understory percent cover and height were estimated in two, 1 m radius plots along each transect. Stump height, diameter, and decay class were measured on all odd numbered plots within the entire 0.04-ha plot.

3.2.4 Allometric biomass estimation

We used species-specific allometric equations developed in the interior of British Columbia by Standish et al. (1985) that utilize measured diameter and height to estimate whole-tree aboveground biomass. Per-tree biomass was then summed for each plot and expressed on an area basis. Initial data examination suggested the Standish et al. (1985) regressions may overestimate small Douglas-fir trees, due to a large intercept term (61.9 kg). Yet, alternative equations developed by Brown (1978) for small trees were parameterized using trees less than 4.6 m in height, limiting the scope of their application for this study. Since neither set of regressions fully addressed all trees less than 10 cm dbh, we tested for differences between the two equations. Ultimately, there was little difference in total tree biomass estimates between

equations and none of the relationships between treatments were altered. Hence, we opted to use Standish et al. (1985) equations for consistency with other biomass calculations used.

Standing dead tree (snag) biomass was calculated using the same Standish et al. (1985) equations, but was adjusted for wood decay using species-specific dead:live biomass density ratios developed by Cousins et al. (2015). The decay classes (e.g. 1-5) in Cousins et al. (2015) were designated as either sound (1-3) or rotten (4 and 5). The mean of the sound ratios (0.92 for ponderosa pine, 0.67 for Douglas-fir) was applied to all standing snags with intact tops, while the average of the higher decay class ratios (0.58 ponderosa pine; 0.51 Douglas-fir) was applied to all snags with broken tops. Whole-tree biomass equations were used for both live and dead trees with broken tops. While this likely underestimates the biomass in these trees given the taper assumptions associated with whole-tree allometries, there are currently no equations addressing trees with broken tops, and the frequency of these trees in our study was low.

For all seedlings (<1.4 m height) , biomass was estimated from height via height-dependent equations developed in western Montana by Brown (1978), who generated whole tree equations for all trees less than 4.6 m tall. Per-seedling biomass was then summed to the plot level and expressed on an area basis.

Aboveground stump biomass was estimated using species-specific stump equations generated by Woodall et al. (2010). Volumes of stumps inside and outside of bark were calculated from top height diameters and bark thickness using equations by Raile (1982). As no stump volume estimators currently exist for western conifer species, red pine (*Pinus resinosa* Aiton) parameters were applied to estimate both ponderosa pine and Douglas-fir stump volumes. These were then adjusted for differences in basic specific gravity of wood among the species

(Woodall et al. 2010). Since species was unidentifiable for the majority of stumps, this parameter was assigned by determining the relative proportion of ponderosa pine and Douglas-fir that were harvested (given pre-harvest and post-harvest species compositions), and then randomly allocating species to stumps on that proportional basis. To account for decay, the same Cousins et al. (2015) dead:live ratios that were applied to snags were also applied to stumps on the basis of recorded stump decay class (S, R).

Forest floor biomass was calculated via planar-intercept sampling (Lutes et al. 2006). Understory vegetation biomass was calculated using the surface fuels-veg equation available in the BIOPAK module of the FIREMON fire effects monitoring and inventory system, where biomass is calculated as a function of height, percent cover, and bulk density (Means et al. 1996, Caratti 2006, Lutes et al. 2006). Bulk density was assigned using composite values from multiple sources in FIREMON: 0.8 kg m^{-3} for herbaceous plants and 1.8 kg m^{-3} for shrubs (Caratti 2006, Lutes 2016).

3.2.6 Data analysis

Ecosystem biomass components were divided into categories for analysis based on similar published studies (Boerner et al. 2008, Finkral and Evans 2008, Campbell et al. 2009, North et al. 2009, Stephens et al. 2009, Sorensen et al. 2011). Tree biomass consisted of all trees greater than breast height (1.4 m), while regeneration biomass was all trees less than or equal to breast height. Forest floor biomass—duff, litter, fine woody debris (FWD), and coarse woody debris (CWD)—were analyzed both separately and pooled. Understory biomass, stump biomass, and snag biomass were analyzed individually. Total aboveground biomass was calculated by summing the biomass of all live and dead components.

All biomass component distributions were severely non-normal, often demonstrating heavy right skewedness, and residuals demonstrated strong deviations from normality using a preliminary mixed-effects model. Log transformations were not feasible for the forest floor biomass estimates due to presence of zero values, and exponential transformations were unable to normalize the residuals. Therefore, non-parametric permutation testing was chosen to analyze differences in biomass levels using several linear contrasts (Manly 2002). First, either an analysis of variance or simple linear model was executed, with biomass as the response and treatment as the explanatory variable. Biomass observations were then randomly reassigned in each unit across all 12 units 999 times to create a sample of F-statistics (or t-statistics) that represent randomly allocated biomass levels. To test the null model (e.g. biomass is the same across treatments), the F-statistic from the original model was compared to the distribution of simulated F-statistics. If the F-statistic was unusually small (less than the 10th percentile; $p < 0.1$) relative to the simulated F-statistics, then those quantities of biomass are unlikely to have arisen if the null model is true, supporting an alternative hypothesis.

This application of permutation testing was used to examine several contrasts for each ecosystem component and the total aboveground biomass. First, an analysis of variance was performed to determine (1) whether biomass in any of the treatments was different than any of the other treatments (CO-NB-SB/WB-FB/DB). Then simple linear modeling was applied to: (2) test whether treating a stand resulted in lower biomass irrespective of treatment type (CO-NB & SB/WB & FB/DB); (3) compare biomass in each of the treatments to the control (CO-NB, CO-SB/WB, CO-FB/DB); (4) contrast each of the burn treatments to the no burn treatment (NB-SB/WB, NB-FB/DB); (5) determine whether biomass differed between the burned treatments versus the no burn treatment (NB-SB/WB & FB/DB); and (6) check for differences in biomass

between the spring/wet burns and fall/dry burns (SB/WB-FB/DB). Matched pairs t-testing was used to examine differences in live tree biomass between pre-treatment (1991) and post-treatment (1993, 2005, and 2015) sampling events in each treatment. Normality of sample distributions ($n = 3$) for each treatment was confirmed via a Shapiro-Wilk normality test.

3.3 Results

The proportional distribution of biomass among components was strikingly similar across all units, regardless of study or treatment. The greatest proportion of aboveground biomass was stored in trees, accounting for 93-95% of the aboveground biomass profile in both the thinning and shelterwood treatments. Of the remainder, approximately 2-3% consisted of forest floor biomass: litter, duff, FWD, and CWD. Snags and stumps each accounted for 1-2%, regeneration accounted for 0.01-0.3%, and understory vegetation accounted for just 0.03-0.08%.

3.3.1 Trees

In both the thinning and shelterwood, pre-treatment (1991) tree biomass was the same across each of the fuel reduction treatments (Table 2), illustrating homogeneity of the forest among designated treatment units. Harvesting removed an average of 28.5 Mg ha⁻¹ of biomass from the thinning treatments and an average of 71.9 Mg ha⁻¹ from the shelterwood (Table 2). In 1993, a year after harvesting, biomass in the thinning treatments was approximately 67% of pre-treatment biomass, and 57% of biomass in the CO. However, there was no difference among fuel reduction treatments, indicating that burning did not immediately further reduce live tree biomass. By 2005, or 13 years after treatment, biomass across all fuel reduction treatments in the thinning had recovered to pre-treatment levels. By 2015, trees in the fuel treatments stored an average of 127% of pre-treatment biomass (Figure 2a). In 2015, biomass levels remained similar

across fuel treatments, while all three were less than the control by an average of 27% ($p = 0.02$, 0.01, and 0.02 for NB, WB and DB respectively) (Figure 3a).

In the shelterwood, average post-treatment (1993) tree biomass in treatments was 47% of the pre-treatment biomass and 40% of the biomass in the CO, but also did not differ among fuel reduction treatments (Table 2). In 2005, biomass in the two burned treatments was still significantly lower than pre-treatment biomass levels. However, biomass in the NB treatments had recovered 2005, although it remained lower than the CO (Figure 2b). By 2015, biomass in the NB was 153% of pre-treatment levels, while in the burned treatments, biomass was only 85 and 89% of pre-treatment biomass in the WB and DB units, respectively (Table 2). Burning (WB & DB) resulted in a little more than half of the biomass accumulation of unburned treatments (NB & CO), even 23 years after treatment (Figure 4a).

Table 2. Mean values of tree biomass (Mg ha^{-1}) pre-treatment (1991), removed from treatment, post-treatment (1993), 13 years after treatment (2005), and 23 years after treatment (2015). * indicates significant differences from pre-treatment (1991) biomass levels at the 0.1-level; ** at the 0.05-level; *** at the 0.01-level.

Thinning					
	1991	Harvested	1993	2005	2015
CO	-	-	102.1	142.0	149.4
NB	88.1	31.9	56.2 ***	86.2	110.2
SB	77.5	23.2	54.3 **	78.8	104.7
FB	93.1	30.4	62.8 **	88.4	112.4
Shelterwood					
	1991	Harvested	1993	2005	2015
CO	-	-	156.7	193.6	214.4
NB	146.8	76.8	70.0 **	140.6	224.8
WB	128.9	69.7	59.2 **	72.4 *	109.7
DB	132.2	69.2	63.0 **	79.2 *	117.9

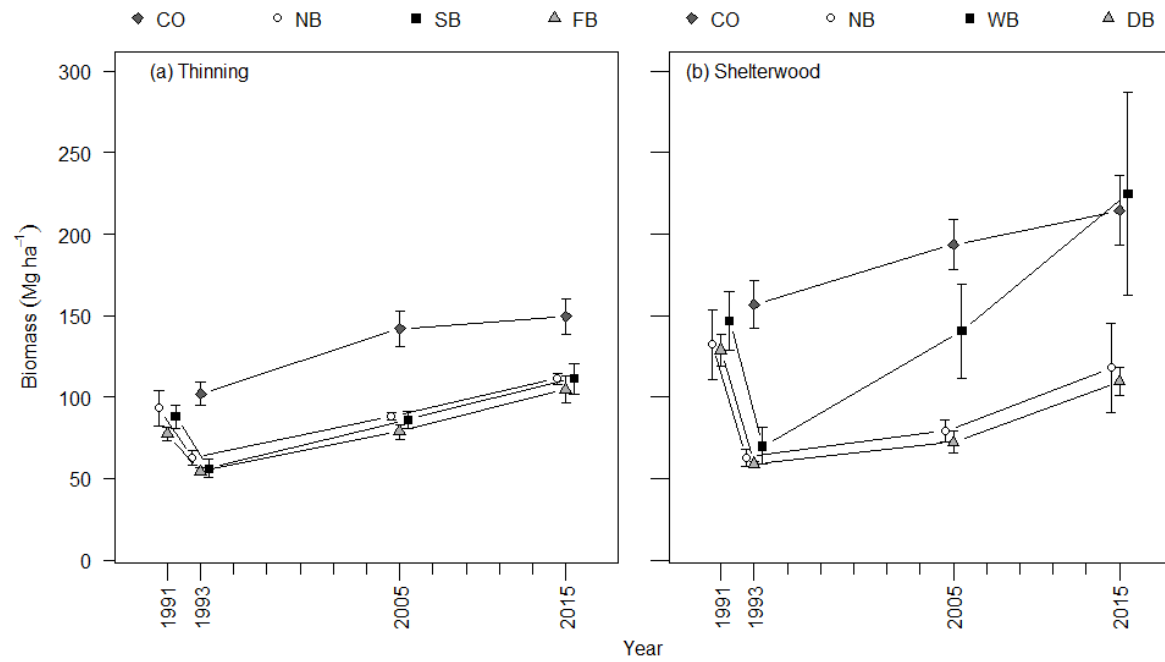


Figure 2. Tree biomass over time, beginning pre-harvest in 1991. First post-treatment remeasurement (1993) was one year after harvesting. Control data was not collected until 1993.

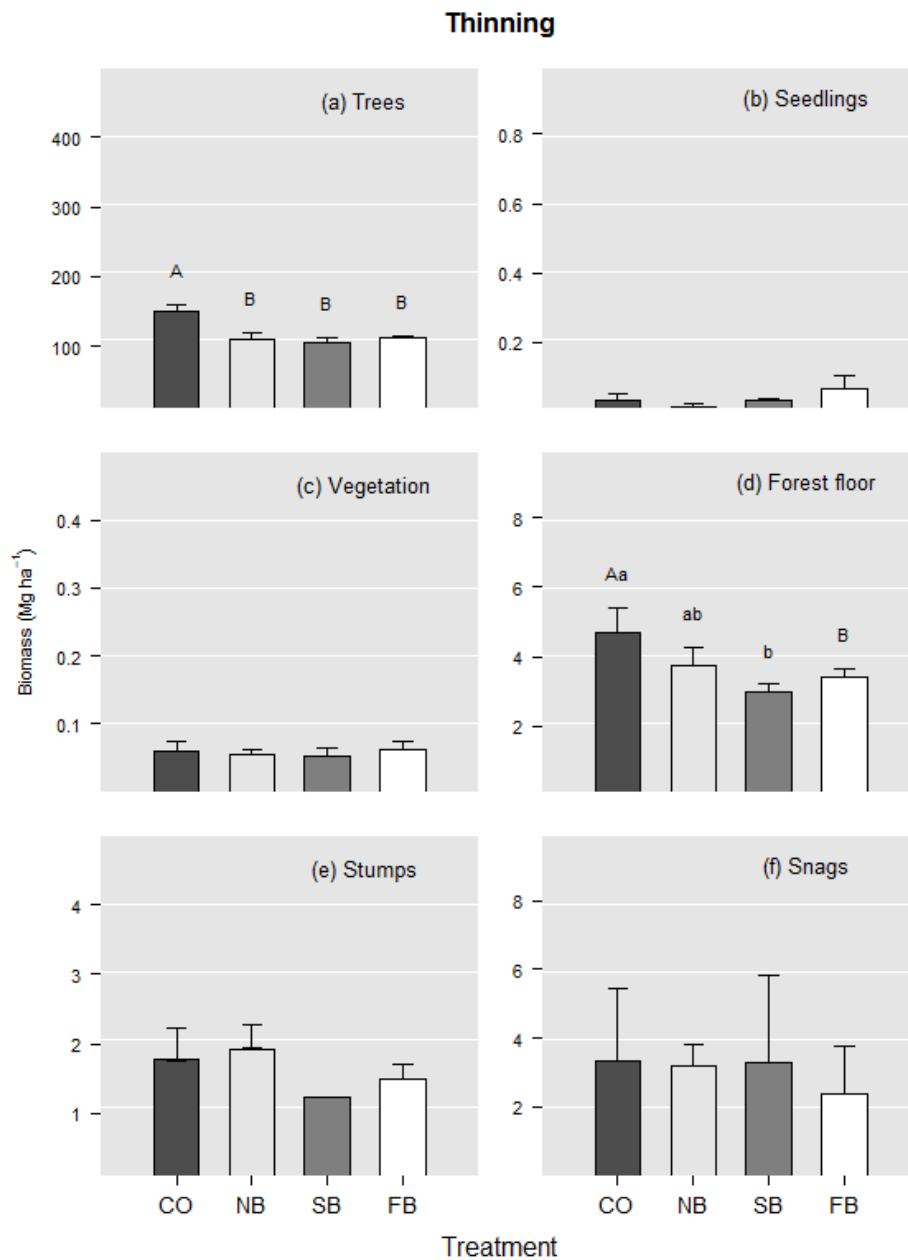


Figure 3. 2015 average biomass in each of the treatment types (control (CO), no burn (NB), spring burn (SB), fall burn (FB)) for each component in the thinning. Trees (a) include all trees ≥ 1.4 m tall, while seedlings (b) are all trees < 1.4 m tall. Vegetation (c) includes all shrubs and herbs. Surface (d) refers to the sum of duff, litter, fine woody debris and coarse woody debris biomass. Stumps (e) are all dead tree remnants < 3 m tall, while snags (f) are dead trees ≥ 3 m tall. Error bars are one standard error from the mean. Letters above whiskers denote significant differences among treatments: uppercase indicates differences at the 0.05-level, while lowercase represents significance at the 0.1-level. Note that biomass scales are not the same for each component.

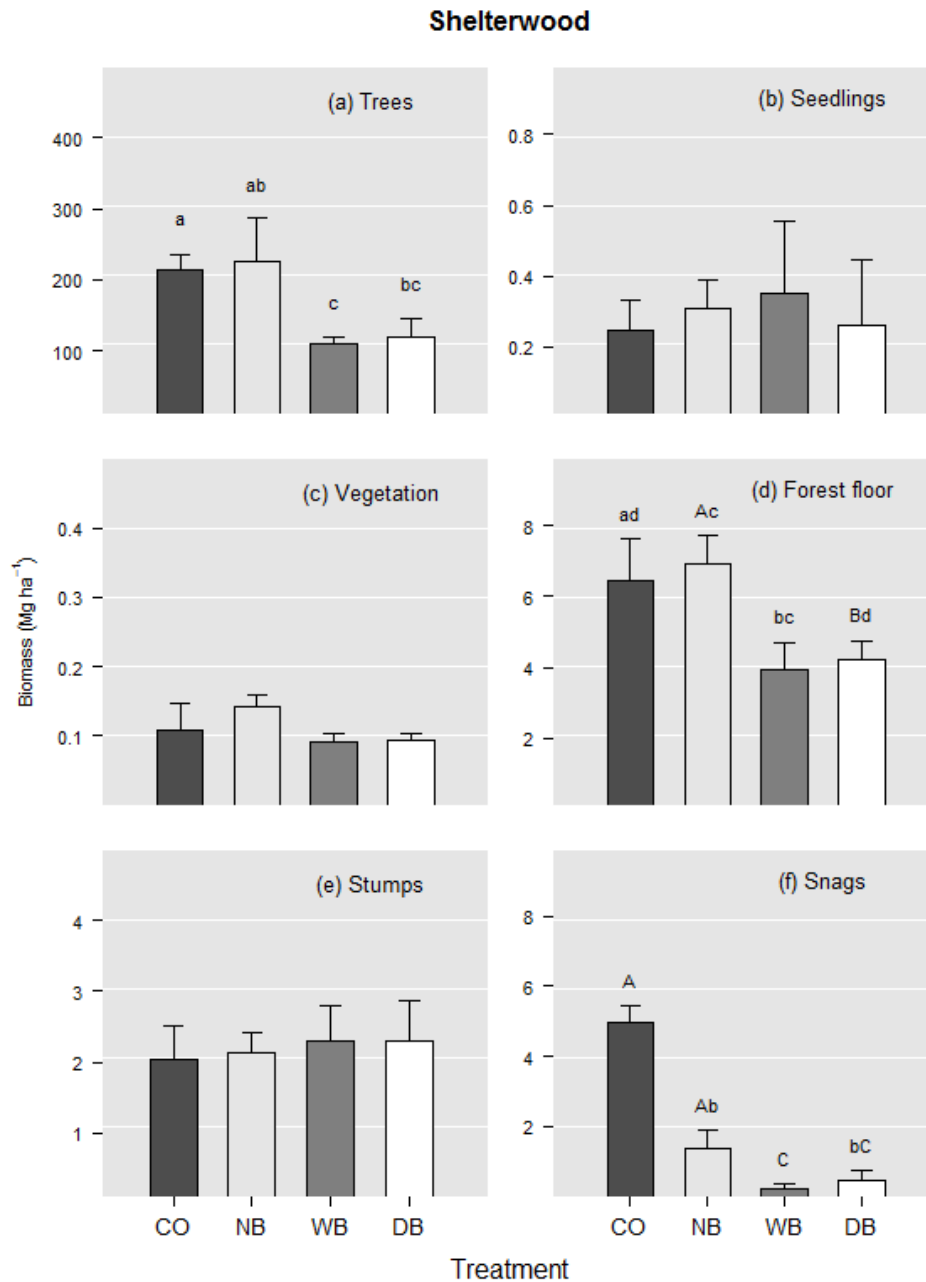


Figure 4. 2015 average biomass in each of the treatment types (control (CO), no burn (NB), wet burn (WB), dry burn (DB)) for each component in the shelterwood. Trees (a) include all trees ≥ 1.4 m tall, while seedlings (b) are all trees < 1.4 m tall. Vegetation (c) includes all shrubs and herbs. Surface (d) refers to the sum of duff, litter, fine woody debris and coarse woody debris biomass. Stumps (e) are all dead tree remnants < 3 m tall, while snags (f) are dead trees ≥ 3 m tall. Error bars are one standard error from the mean. Letters above whiskers denote significant differences among treatments: uppercase indicates differences at the 0.05-level, while lowercase represents significance at the 0.1-level. Note that biomass scales are not the same for each component.

3.3.2 *Forest floor*

In 2015, differences in forest floor biomass in the thinning were primarily due to lower levels of duff and litter biomass in the fuel reduction treatments relative to the control. Thinned stands had between 15 to 23% less litter and 43 to 74% less duff than the CO, with the greatest reductions in the burned treatments (Figure 5). Total forest floor biomass ranged from 3 Mg ha⁻¹ in the WB to 4.7 Mg ha⁻¹ in the CO (Table 3). Thinning reduced forest floor biomass by 20% relative to the CO, while thinning and burning reduced forest floor biomass by an additional 8 to 16% (FB, SB, respectively) (Figure 3d). In the shelterwood, burning resulted in lower levels of CWD, FWD and duff biomass compared to both the CO and NB, reducing total forest floor biomass by 34 to 39% relative to the CO and by 39 to 43% relative to the NB (Figure 6). Differences in conditions during the burns were apparently unimportant as there was no difference in forest floor biomass between the SB and FB in the thinning or the WB and DB in the shelterwood.

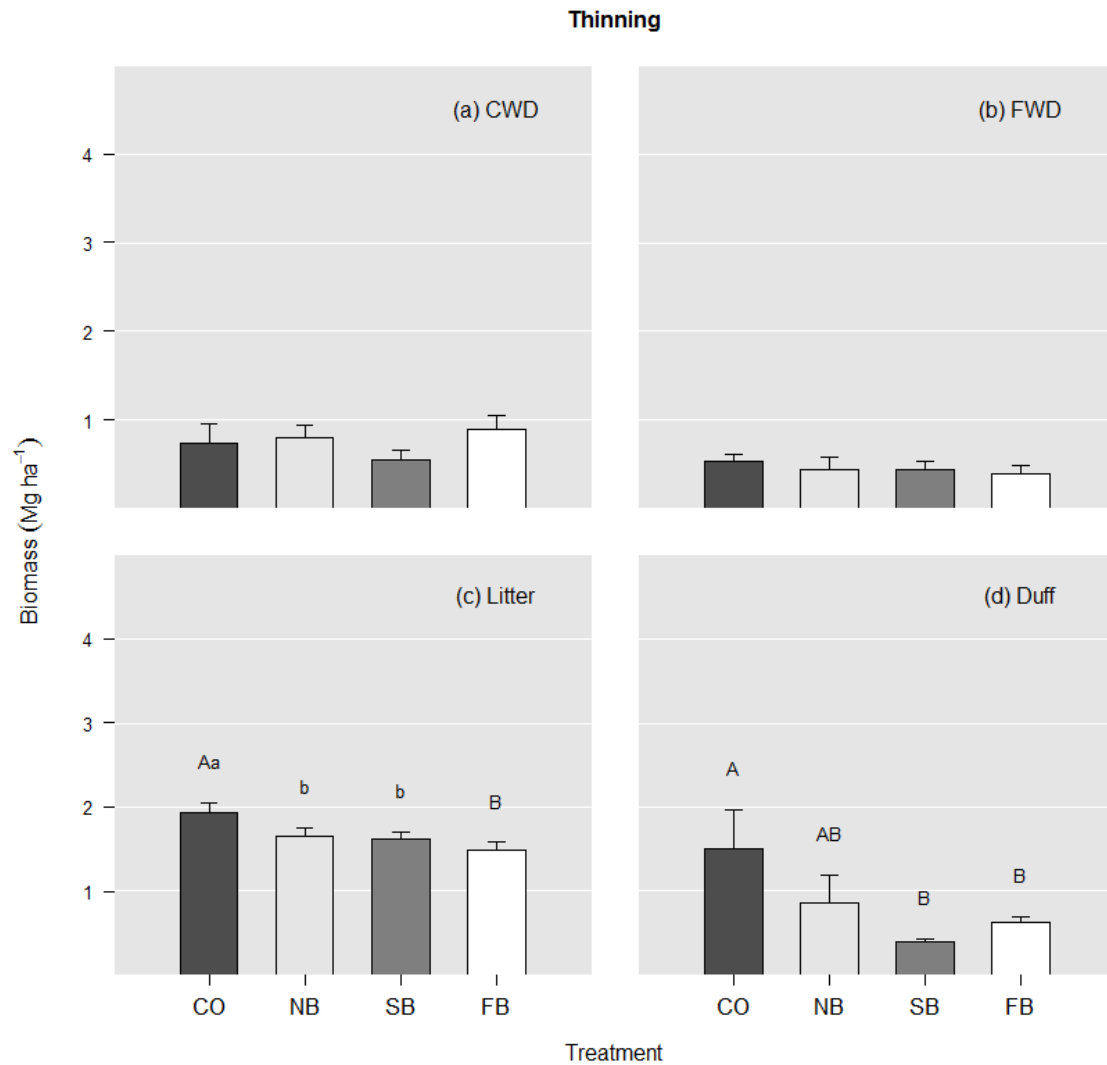


Figure 5. 2015 forest floor biomass components: coarse woody debris (a), fine woody debris (b), litter (c) and duff (d) in the thinning for each of the four treatment types (control (CO), no burn (NB), spring burn (SB), fall burn (FB)). Alphabetical letters indicate significance at the 0.05-level (uppercase) and 0.1-level (lowercase).

Table 3. Mean values of biomass (Mg ha^{-1}) 23 years post-treatment for all ecosystem components in each treatment (standard errors in parentheses). Treatments are control (CO), no burn (NB), spring or wet burn (SB/WB) and fall or dry burn (FB/DB). Significant relationships within rows are denoted using alphabetical letters, with capital letters indicating significance at the < 0.05 level and lowercase letters representing significance at the < 0.1 level.

	Thinning											
	CO			NB			SB			FB		
Trees	149.38	(10.77)	A	110.24	(9.51)	B	104.66	(7.98)	B	112.43	(2.52)	B
Seedlings	0.03	(0.02)		0.02	(0.01)		0.03	(0.01)		0.07	(0.04)	
Vegetation	0.06	(0.01)		0.05	(0.01)		0.05	(0.01)		0.06	(0.01)	
CWD	0.73	(0.22)		0.80	(0.14)		0.55	(0.10)		0.89	(0.16)	
FWD	0.53	(0.08)		0.45	(0.14)		0.44	(0.09)		0.39	(0.08)	
Litter	1.51	(0.46)	Aa	0.87	(0.32)	b	0.39	(0.05)	b	0.63	(0.07)	B
Duff	1.51	(0.46)	A	0.87	(0.32)	AB	0.39	(0.05)	B	0.63	(0.07)	B
Stumps	1.75	(0.48)		1.93	(0.33)		1.22	(0.01)		1.51	(0.21)	
Snags	3.35	(2.10)		3.23	(0.63)		3.30	(2.53)		2.41	(1.36)	
Total	159.27	(8.34)	A	119.23	(8.55)	B	112.26	(6.21)	B	119.88	(2.56)	B

	Shelterwood											
	CO			NB			WB			DB		
Trees	214.44	(21.43)	a	224.83	(62.06)	ab	109.68	(8.70)	c	117.87	(27.59)	bc
Seedlings	0.25	(0.09)		0.31	(0.08)		0.35	(0.20)		0.26	(0.19)	
Vegetation	0.11	(0.04)		0.14	(0.02)		0.09	(0.01)		0.09	(0.01)	
CWD	2.33	(0.68)	a	2.58	(0.32)	ab	1.12	(0.27)	b	1.25	(0.35)	ab
FWD	0.84	(0.19)	ab	0.91	(0.10)	a	0.49	(0.09)	b	0.67	(0.12)	ab
Litter	1.50	(0.12)		1.64	(0.19)		1.51	(0.43)		1.50	(0.09)	
Duff	1.79	(0.22)	A	1.80	(0.42)	AB	0.81	(0.18)	C	0.82	(0.16)	BC
Stumps	2.03	(0.46)		2.12	(0.28)		2.29	(0.49)		2.28	(0.57)	
Snags	4.99	(0.45)	A	1.38	(0.54)	B	0.26	(0.12)	B	0.48	(0.27)	B
Total	228.28	(23.16)	a	235.73	(61.79)	ab	116.60	(9.36)	c	125.23	(28.64)	bc

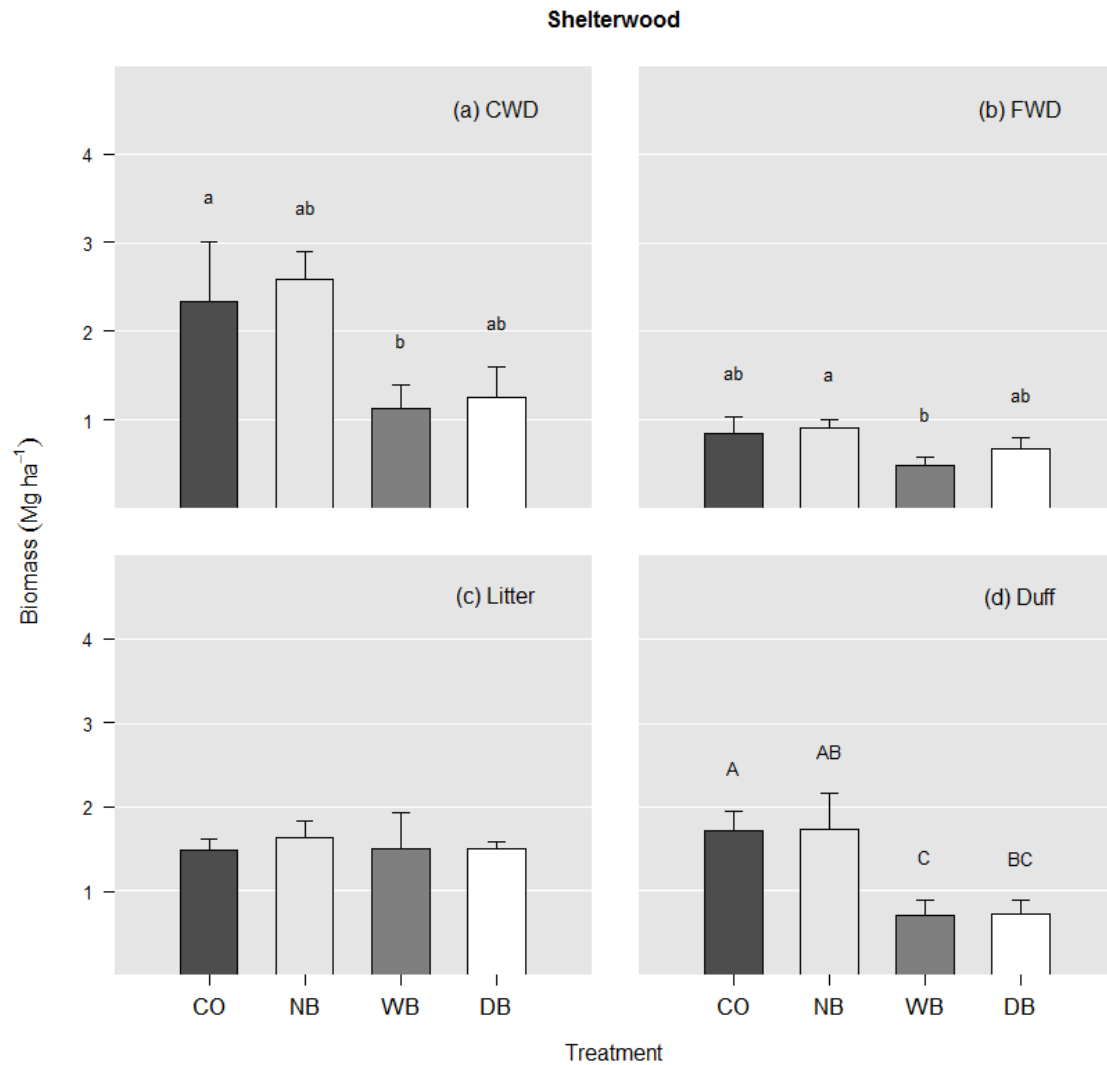


Figure 6. 2015 forest floor biomass components by coarse woody debris (a), fine woody debris (b), litter (c) and duff (d) in the shelterwood for each of the four treatment types (control (CO), no burn (NB), wet burn (WB), dry burn (DB)). Alphabetical letters indicate significance at the < 0.05 level (uppercase) and < 0.1 level (lowercase).

3.3.3 Stumps and Snags

Stump biomass levels were similar across all treatments for both the thinning ($p = 0.44$) and the shelterwood ($p = 0.95$); this included even the controls, due to the ubiquitous presence of stumps from past harvesting in the area (Figure 3e, 4e). Snag biomass was homogeneous across all treatments in the thinning study ($p = 0.99$) (Figure 3f). However, in the shelterwood study,

snag biomass in the CO was between 4 to 20 times greater than in each of the fuel treatments ($p = 0.01$ for NB, $p \sim 0.00$ for WB, DB), likely as a result of competition (Figure 4f). Snag biomass was lowest in the burn treatments, and burning further reduced snag biomass relative to the NB treatment by as much as 84% ($p = 0.04$).

3.3.4 Understory Vegetation and Seedlings

In 2015, understory vegetation biomass was the same across all treatments in both the thinning ($p = 0.95$) and shelterwood ($p = 0.44$) (Figure 3c). In the thinning, vegetative biomass ranged from 0.05 Mg ha^{-1} (NB and SB) to 0.06 Mg ha^{-1} (CO and FB) (Table 3). Similarly, in the shelterwood, biomass ranged from 0.09 Mg ha^{-1} (WB and DB) to 0.14 Mg ha^{-1} (NB) (Figure 4c). Seedling biomass was also similar across all treatments, including the control, in both the thinning ($p = 0.51$) and shelterwood ($p = 0.96$) (Figure 3b, 4b). In the thinning, seedlings accounted for 0.02 Mg ha^{-1} (NB) to 0.07 Mg ha^{-1} (FB) (Table 3). In the shelterwood, seedling biomass was much greater, accounting for 0.25 Mg ha^{-1} (CO) to 0.35 Mg ha^{-1} (WB).

3.3.5 Total aboveground biomass

Total biomass 23 years after treatment in the thinning was significantly lower in each of the three fuel reduction treatments (NB, SB, FB) by 25-30% relative to the control's 159.7 Mg ha^{-1} ($p < 0.01$ for all three treatments) (Figure 7). There was no difference among the three fuel reduction treatments, with the NB units storing an average of 119.9 Mg ha^{-1} of biomass, the SB units an average of 112.2 Mg ha^{-1} , and the FB units an average of 119.2 Mg ha^{-1} (Table 3). In the shelterwood, burning reduced total biomass by 50% in the WB and 45% in the DB relative to the CO ($p = 0.08$ for WB and DB each) (Figure 7). The NB treatment however, resulted in

similar levels of aboveground biomass as the CO (228 Mg ha⁻¹), with an average of 235.7 Mg ha⁻¹. The WB (116.6 Mg ha⁻¹) and DB (125.2 Mg ha⁻¹) mean biomass levels were a little more than half the biomass in the NB, but only the difference between the WB and NB was significant ($p=0.05$) (Table 3). There was no difference in biomass between the two types of burning ($p = 0.85$).

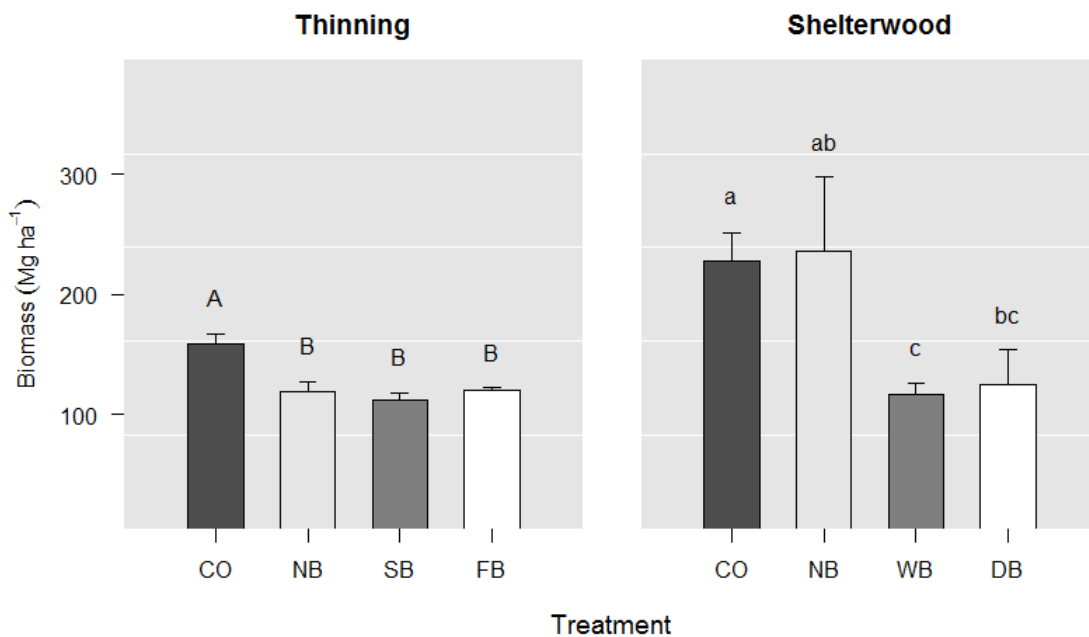


Figure 7. Total aboveground biomass by treatment for each silvicultural prescription in 2015, 23 after harvesting. Error bars represent one standard error from the mean. Alphabetical letters indicate significant differences: uppercase letters represent significance at the <0.05 -level, while lowercase letters suggest significance at the < 0.1 -level.

3.4 Discussion

There were several notable long-term responses of biomass components to treatments. Primarily, our results indicate that tree biomass can return to pre-harvest levels in less than 13 years in some cases, and by 23 years in others, although they remained below 2015 control levels. Furthermore, at least in the thinning, recovered tree biomass was stored in fewer trees,

indicating that residual biomass is stored in larger trees. For example, densities in fuel treatments in the thinning ranged from 29 (NB) to 117 (FB) fewer TPH in 2015 than in 1991, but biomass levels were the same between years. In the shelterwood, tree biomass in the two burned treatments were just approaching pre-treatment levels, while the NB treatment resulted in even greater amounts of biomass than before treatment. This additional tree biomass in the NB units can be explained by differentiating between saplings (<10 cm dbh) and overstory trees across the three fuel reduction treatments. While the burned treatments only had between 23.5 Mg ha⁻¹ (WB) and 30.3 Mg ha⁻¹ (DB) of biomass stored in saplings, the NB treatment had 118.0 Mg ha⁻¹ of biomass stored in saplings, more than half of the entire tree biomass. These values reflect the development of advanced regeneration that arose in the absence of burning treatments. This is similar to a study examining the effect of various combinations of thinning and burning on rates of biomass accumulation 8 years after treatment in Sierra Nevada mixed-conifer forests. Hurteau and North (2010) observed that the greatest rate of live-tree carbon accumulation, as well as overall live-tree carbon storage, was accomplished using an understory thinning without burning. In that study, burned units experienced almost half of the carbon gains in large trees over the same duration of time. This rapid development of tree biomass was not observed in the NB treatment in the thinning study at Lick Creek, likely because of lower site productivity. The shelterwood installation is located in a more productive location at the base of the drainage in the vicinity of mesic species, including grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.), Engelmann spruce (*Picea engelmannii* Parry ex Engelman), and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*).

In both studies, it appears that even 23 years after treatment, cutting and burning treatments continue to maintain lower levels of forest floor biomass than either the untreated or

no burn stands, contrary to our initial hypothesis. In the thinning study, lower fuel biomass was attributed entirely to duff and litter, which was lower in each of the two burned treatments relative to the control. Litter was also suggestively lower in the NB units relative to the control. These results indicates that litter decomposition rates may be primarily benefiting from sustained exposure to sunlight in the all of the thinned units, regardless of underburning. Reduced amounts of duff however, still appears to be an artifact of the initial underburning, even 23 years after treatment. In the shelterwood, only duff was lower for both burned treatments relative to the control. CWD was also suggestively lower in the shelterwood WB relative to the control, but not the DB. The greatest differences in woody biomass in the shelterwood were between the NB and burned units, as duff, FWD, and CWD were all lower in the burned units when the WB and DB were pooled. Again, this can likely be attributed to the expansion of ingrowth into the overstory, increasing sources for woody debris, as well as lessened microbial decomposition activity from decreased exposure to sunlight. Shorter-term studies have detected an initial increase in forest floor biomass, ranging from 18-28% more litter and fine woody debris after thinning (Stephens and Moghaddas 2005a, North et al. 2009), as well as decreases, ranging from 10-24% (North et al. 2009, Sorensen et al. 2011). Several studies detected no change at all (Boerner et al. 2008, Finkral and Evans 2008, Stephens et al. 2009). When broadcast burning was added to the treatment, short-term studies almost unanimously observed a decrease in fine surface biomass (Stephens and Moghaddas 2005a, Boerner et al. 2008, North et al. 2009, Stephens et al. 2009). This is to be expected, as these fine, relatively dry, woody biomass materials are the primary source of fuels carrying surface fires.

In our study, snag volumes in treated stands were either the same as controls (thinning study) or drastically decreased relative to the control, regardless of underburning (shelterwood

study). Mortality in both installations seemed to be driven by competitive stress and mountain pine beetle (*Dendroctonus ponderosae*), with some presence of comandra blister rust (*Cronartium comandrae* Pk.). Other studies have observed an initial increase or no difference in the volume of snags after thinning relative to the control (Stephens and Moghaddas 2005b, Boerner et al. 2008, North et al. 2009), where mortality be a result of potential logging damages during harvesting. In thinning and burning treatments, snag biomass has shown mixed responses, seemingly dependent on burn intensity (Stephens and Moghaddas 2005b, North et al. 2009).

Our study indicates that by 23 years after treatment, understory vegetation no longer benefits from fuel treatments, and is homogeneous across treatment types. In a shorter-term study in a Sierra Nevada mixed-conifer forest, understory biomass increased almost threefold in the thinning treatment relative to the control within the three years following treatment (Campbell et al. 2009). When fuel treatments in fire-frequent forest types involves canopy removal, understory production typically increases with improved access to sunlight and belowground resources (Connell and Smith 1970, Campbell et al. 2009). However, as the overstory canopy recovers over time, this advantage to understory vegetation diminishes, and in the long term, we expect vegetation to be similar across treatments, as observed in our study.

Several recommendations were made by Galik and Jackson (2009) on considerations for maximizing carbon storage while minimizing risk of offset reversal from wildfire. Proposals relevant to mixed and pure ponderosa pine forests to maximize carbon storage are (1) lengthening thinning rotations, (2) incorporating mixed species and mixed age stands, and (3) promoting thinning regimes with higher residual stem densities. These suggestions must be balanced against procedures for reducing wildfire severity, which typically entail shortened thinning rotations, prioritizing shade-intolerant species and reducing ladder fuels created by

mixed-aged structures, and substantially decreasing stem densities (Galik and Jackson 2009). In the context of fuel reduction efforts, treatments that prolong the amount of time required before another entry can help reduce carbon emissions from harvesting operations or burning, as well as increase carbon sequestration of the stand. At Lick Creek, all three fuel reduction treatment options continued to perpetuate forest structures conducive to low-severity fires on the low-productivity site (thinning study), while only the two treatments that included underburning are still effective on the high productivity site (shelterwood). Both thinning and shelterwood prescriptions promote differentiation of age classes, and to some extent, mixed-species. If the management objective is to promote fire resilient forest structure to lessen offset potential from fire-induced mortality, species compositions with higher proportions of ponderosa pine are preferred, such as the SB and CO in the thinning, and DB and WB in the shelterwood. Finally, stem densities in the thinning study were lowest in the FB (269 trees per hectare), intermediate in the NB and SB (356 and 362 TPH), and highest in the CO (566 TPH). These results might indicate that the NB and SB treatments are effectively balancing carbon storage while maintaining fire resilient structures. In the shelterwood, stems per hectare were lowest in the two burned treatments (825 in DB, 853 in WB), slightly higher in the CO (1036 TPH), and immensely greater in the NB (2261 TPH). In this case, the NB is sequestering the most carbon but poses a higher risk of offset reversal during wildfire. Furthermore, the high density of ladder fuels in the NB treatment in the shelterwood indicates that the decision not to burn in a high productivity site may substantially reduce treatment longevity, and will require several entries to maintain the same stand structure as a single cut-and-burn treatment application, potentially increasing the carbon cost of maintaining the treatment.

Existing research indicates that carbon emissions from treated stands with similar attributes to the current stand structure of the Lick Creek fuel reduction treatments were significantly lower than untreated stands (e.g. Hurteau and North 2009, Stephens et al. 2009, Reinhardt and Holsinger 2010, Yocom Kent et al. 2015). However, it is unclear as to whether total carbon emissions from thinning, prescribed burning, and the subsequent wildfire will be lower than those of an untreated stand. Future analysis should address this question by modeling carbon emissions from potential wildfires in these and other stands, and determine net carbon loss from treatment and wildfire combinations. Current carbon accounting policies such as the California Climate Action Registry (2007) do not penalize landowners for carbon emissions from wildfires (Hurteau et al. 2008). They do however, consider emissions from fuel treatments a carbon source and deduct credits from these activities, disregarding potential reductions in emissions during future wildfires. Therefore, it is important to understand whether these carbon credit policies are fairly subtracting credits from landowners who take preemptive action. Some studies, such as Hurteau and North (2010), suggested that carbon emitted from prescribed fire would be sequestered by boosted growth of trees and shrubs within a time period shorter than the historic mean fire return interval. However, others indicate that the probability of a wildfire occurring in a treated stand during the span of treatment longevity is negligible, negating any potential reduction in net emissions (Rhodes and Baker 2008, Campbell et al. 2012, North et al. 2012). While our research provides information regarding biomass accumulation in the Northern Rockies, future enquiry should incorporate regional disturbance regimes into analysis of potential emissions tradeoffs.

3.5 Conclusions

The Lick Creek studies analyzed here provide a unique opportunity to examine long-term biomass dynamics and compare treatment responses across two sites with different treatment histories, site productivities, and topographic locations while still holding climate and geographic variables constant. Our results indicated that stands can recover harvested biomass from fuel reduction treatments in as little as 13 years while still achieving restoration objectives, especially if harvesting is followed by prescribed burning. Furthermore, burning resulted in lower levels of forest floor biomass even 20 years after treatment, while results in the no-burn treatment were highly dependent on site productivity. Results from our study suggest a feasible balance between carbon storage and restoration goals in the fire-frequent forests of the northern Rocky Mountains.

Appendices

Appendix A. F-statistics for each contrast tested for each biomass component. P-values from the non-parametric permutation testing are included in parentheses, while stars indicate significance at the 0.01-level (***), 0.05-level (**), and 0.1-level (*)

Thinning									
Contrasts	Surface fuels	Vegetation	Stumps	Seedlings	Snags	Trees		Total	
NB - WB	2.36 (0.13)	0.13 (0.95)	0.94 (0.44)	0.90 (0.51)	0.06 (0.99)	6.02 (0.03)	**	9.73 (0.02)	**
NB & WB	5.75 (0.04)	** 0.04 (0.83)	0.33 (0.60)	0.03 (0.89)	0.03 (0.87)	17.60 (0.01)	***	28.44 (0.00)	***
CO - NB	-1.40 (0.18)	-0.29 (0.77)	0.31 (0.76)	-0.56 (0.62)	0.05 (0.97)	-3.33 (0.02)	**	-4.14 (0.01)	***
CO - WB	-2.55 (0.03)	** -0.39 (0.70)	-1.23 (0.24)	-0.07 (0.96)	-0.02 (0.98)	-3.80 (0.01)	***	-4.86 (0.00)	***
CO - DB	-1.92 (0.08)	* 0.17 (0.87)	-0.63 (0.55)	1.04 (0.31)	-0.37 (0.71)	-3.14 (0.02)	**	-4.07 (0.01)	***
3 - WB & I	0.92 (0.34)	0.04 (0.85)	2.06 (0.17)	1.45 (0.26)	0.03 (0.88)	0.03 (0.88)		0.14 (0.73)	
NB - WB	1.04 (0.33)	0.14 (0.70)	1.67 (0.24)	0.13 (0.77)	0.05 (0.86)	0.43 (0.55)		0.76 (0.41)	
NB - DB	0.27 (0.62)	0.21 (0.66)	0.95 (0.35)	2.56 (0.13)	0.10 (0.74)	0.03 (0.87)		0.00 (0.96)	
WB - DB	0.39 (0.54)	0.31 (0.58)	0.37 (0.55)	1.24 (0.30)	0.12 (0.74)	0.44 (0.54)		0.62 (0.46)	
Shelterwood									
Contrasts	Surface fuels	Vegetation	Stumps	Seedlings	Snags	Trees		Total	
NB - WB	3.31 (0.07)	* 1.08 (0.44)	0.07 (0.95)	0.10 (0.96)	33.34 (0.00)	*** 2.93 (0.12)		3.15 (0.09)	*
NB & WB	2.18 (0.19)	0.00 (1.00)	0.15 (0.71)	0.12 (0.74)	95.09 (0.00)	*** 2.36 (0.18)		2.73 (0.15)	
CO - NB	0.39 (0.70)	1.05 (0.35)	0.14 (0.87)	0.30 (0.78)	-6.70 (0.01)	*** 0.20 (0.85)		0.14 (0.90)	
CO - WB	-2.14 (0.07)	* -0.53 (0.62)	0.38 (0.67)	0.49 (0.63)	-8.80 (0.00)	*** -2.07 (0.09)	*	-2.18 (0.08)	*
CO - DB	-1.88 (0.11)	-0.49 (0.62)	0.38 (0.71)	0.07 (0.94)	-8.38 (0.00)	*** -1.90 (0.09)	*	-2.01 (0.08)	*
3 - WB & I	7.69 (0.03)	** 3.23 (0.12)	0.06 (0.79)	0.00 (0.98)	4.74 (0.04)	** 6.39 (0.04)	**	6.68 (0.03)	**
NB - WB	5.16 (0.05)	** 0.88 (0.38)	0.02 (0.87)	0.13 (0.71)	2.11 (0.17)	5.15 (0.06)	*	5.40 (0.05)	*
NB - DB	2.59 (0.14)	2.35 (0.17)	0.06 (0.81)	0.05 (0.84)	2.81 (0.12)	1.26 (0.30)		1.31 (0.29)	
WB - DB	0.07 (0.78)	0.00 (0.96)	0.00 (0.99)	0.18 (0.70)	0.18 (0.71)	0.03 (0.86)		0.03 (0.85)	

Appendix B. F-statistics for each contrast tested for each of the woody surface biomass components. P-values from the non-parametric permutation testing are included in parentheses, while stars indicate significance at the 0.01-level (***), 0.05-level (**), and 0.1-level (*).

Thinning Surface Fuels										
Contrasts	Duff			Litter			FWD		CWD	
CO - NB - WB - DB	2.88	(0.08)	*	3.17	(0.08)	*	0.32	(0.80)	0.83	(0.49)
CO - NB & WB & DB	7.24	(0.02)	**	8.30	(0.02)	**	0.81	(0.38)	0.00	(0.94)
CO - NB	-1.61	(0.13)		-1.94	(0.09)	*	-0.60	(0.57)	0.31	(0.73)
CO - WB	-2.79	(0.02)	**	-2.14	(0.07)	*	-0.64	(0.54)	-0.83	(0.43)
CO - DB	-2.19	(0.05)	**	-2.97	(0.01)	***	-0.96	(0.37)	0.69	(0.50)
NB - WB & DB	1.05	(0.32)		0.51	(0.48)		0.06	(0.81)	0.19	(0.68)
NB - WB	1.07	(0.31)		0.13	(0.72)		0.02	(0.89)	2.34	(0.17)
NB - DB	0.34	(0.58)		1.07	(0.30)		0.13	(0.72)	0.15	(0.71)
WB - DB	0.37	(0.55)		0.69	(0.43)		0.10	(0.76)	2.29	(0.16)

Shelterwood Surface Fuels										
Contrasts	Duff			Litter			FWD		CWD	
CO - NB - WB - DB	4.61	(0.05)	**	0.08	(0.97)		1.97	(0.18)	2.90	(0.11)
CO - NB & WB & DB	4.55	(0.08)	*	0.04	(0.90)		1.02	(0.35)	0.04	(0.90)
CO - NB	0.02	(0.99)		0.43	(0.66)		0.33	(0.76)	0.41	(0.70)
CO - WB	-2.63	(0.05)	**	0.04	(0.97)		-1.88	(0.11)	-1.96	(0.10) *
CO - DB	-2.61	(0.05)	**	0.01	(0.99)		-0.93	(0.41)	-1.75	(0.13)
NB - WB & DB	9.27	(0.02)	**	0.22	(0.67)		3.98	(0.07) *	6.83	(0.04) **
NB - WB	7.01	(0.04)	**	0.15	(0.72)		4.85	(0.06) *	0.15	(0.72)
NB - DB	2.26	(0.16)		0.07	(0.79)		0.03	(0.85)	0.07	(0.79)
WB - DB	0.00	(0.98)		0.00	(0.97)		0.90	(0.36)	0.05	(0.81)

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